The Ecological Economics of Biodiversity

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2003

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PREFACE

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This book surveys and highlights the potential and limitations of an ecological-economics analysis of biodiversity. It is based on the firm belief that an economic perspective on complex biodiversity issues firmly supported by ecological insights can help to clarify the processes, functions and values associated with biodiversity. This study aims to offer a review of key ecological and economic concepts that are essential in building bridges between ecology and economics, and suggests how to integrate these. Issues discussed include, among others, biodiversity indices, ecosystem management principles and modern ecological concepts such as resilience. Particular attention is given to various monetary valuation approaches and methods from the perspective of conservation and sustainable use of biodiversity. For this purpose, the interrelation between biodiversity, resources, ecosystem functions, and human welfare is examined. The use of ecological and value indicators in integrated modelling and analysis is also addressed. Throughout, several illustrative applications are presented in order to demonstrate the usefulness of the approaches discussed. Ultimately, this study aims to contribute to a better basis for public decision-making regarding biodiversity protection.

The authors would like to acknowledge helpful discussions and exchanges with Professor Yohanes E. Riyanto, Department of Economics, National University of Singapore, Dr Gabriella Vindigni, Faculty of Economics, University of Catania and Dr Dan Biller, World Bank.

Amsterdam, July 2002

The authors

PART I
Context

PREFACE

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This book surveys and highlights the potential and limitations of an ecological-economic analysis of biodiversity. It is based on the firm belief that an economic perspective on complex biodiversity issues firmly supported by ecological insights can help to clarify the processes, functions and values associated with biodiversity. This study tries to offer a review of key ecological and economic concepts that are essential in building bridges between ecology and economics, and suggests how to integrate them. Issues discussed include, among others, biodiversity indices, ecosystem management principles and modern ecological concepts such as resilience. Particular attention is given to various monetary valuation approaches and methods from the perspective of conservation and sustainable use of biodiversity. For this purpose, the relationships between biodiversity, ecosystems, ecosystem functions and human welfare is examined. The use of ecological and value indicators in integrated modelling and analysis is also addressed. Throughout, several illustrative applications are presented in order to demonstrate the usefulness of the approaches discussed. Ultimately, this study aims to contribute to a better basis for public decision-making regarding biodiversity protection.

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Chapter 1
OVERVIEW

1.1 Setting the scene

The worldwide decay in environmental quality and the gradual depletion of natural resources, sometimes referred to as the 'new scarcity', has prompted an intense scientific attention in both the natural and social sciences. The global interest in environmental-economic matters is partly caused by the increased pressure that mounting population and increased production exert on the earth's natural resource base. In addition, as personal incomes rise and leisure time becomes more freely available in the developed world, concern for more immediate human needs has been accompanied by interest in nature preservation and conservation for future generations. Although resource and environmental issues manifest themselves on local or regional scales, they are part of a globally interwoven ecosystem. Consequently, the 'new scarcity' has spatial and temporal horizons that extend far beyond the current level of thinking and acting.

Against this background, the concept of sustainable development has gained much popularity. It was officially proposed as a policy goal in the publication *Our Common Future* by the World Commission on Environment and Development (Bruntland 1987). The Commission called attention to the need to consider our planet as an integrated social, economic, ecological and political system that requires collective initiatives in order to ensure continuity under changing conditions. The report suggested that economic growth and environmental protection can go together and are not necessarily mutually antagonistic forces. This hypothesis has induced much theoretical and empirical research and has led many applications, in particular in the field of the so-called 'green Kuznets curve' (see for an extensive review de Bruyn 1999). The idea of compatibility between growth and the environment was also critically investigated by Duchin *et al.* (1994). Using a multisector-multiregion model for the world, they found that economic growth and environmental quality are in conflict, given certain expectations about technological change.

A dominant element in recent discussions about sustainable development is worry about the loss of biological diversity (or biodiversity). Biodiversity requires our attention for two reasons. First, it provides a wide range of direct and indirect benefits to mankind, which occur on both local and global scales. Second, many human activities contribute to unprecedented rates of biodiversity loss, which threaten the stability and continuity of ecosystems as well as their provision of goods and services to mankind. Consequently, in recent years much attention has been directed towards the analysis and valuation of the loss of biodiversity.

The valuation of biodiversity loss can be approached from an ecological, economic or combined perspective. The present study addresses all three options. This includes attention to the ecological and economic foundations of biodiversity analysis and valuation. Relevant concepts and valuation methods are identified and discussed. In addition, empirical applications are reviewed. Finally, the study addresses multidisciplinary modeling. This allows for the description of the complexity of ecosystem functions and processes, which can be integrated in a transparent way with economic valuation results.

In order to arrive at this stage, a number of biological, ecological and economic issues and questions need to be dealt with. For example, what are the implications of biodiversity for the structure and functions of ecosystems? What underlying driving forces influence the loss of biodiversity? What direct and indirect roles does biodiversity have for human society? What considerations are relevant in making decisions about the conservation of biodiversity? These are important questions that will guide the present study.

Valuation and indicator information play a crucial role in assisting policymakers in the design of resource reallocation plans, contributing to ensure the sustainable use of biodiversity. Available ecological and economic methods for generating empirical indicators are discussed, along with characteristics of empirical applications. It will be noted that it is unclear whether available studies always specifically address biodiversity. The reason is that biodiversity is often associated with complex ecosystem functions and processes that relate only very indirectly to human welfare. As a result, 'resource valuation' and 'biodiversity valuation' are often confused. Thus there is a clear reason for a more specific approach. Biodiversity indicator and valuation techniques will be reviewed here with a focus on providing guidance about how to select the method that is most suitable for assessing a particular, desired component of biodiversity value.

1.2 A historical perspective on the economy-environment relationship

Interest in biodiversity issues initially developed in the 1970s when the growing recognition of worldwide environmental decay and severe resource depletion, fuelled by a population explosion, received unexpected but welcome support during the oil crisis. This sudden event was complemented by the First Report to the Club of Rome, 'The limits to growth', which was based on a scenario analysis with a systems dynamics model of the world. Although the scientific contents of the study left much to be desired – as a result of conservative estimates of resource availability and insufficient incorporation of behavioral and technological feedback mechanisms in the model – it created a shock effect among social scientists. This gave rise to an intensive debate between growth optimists and pessimists (Daly and Townsend 1993; van den Bergh and de Mooij 1999).

It also marked the beginning of the social science interest in environmental problems. Economists constructed abstract models of economic growth and resource use (Dasgupta and Heal 1979) and developed a theory of environmental policy for correcting environmental externalities (Baumol and Oates 1988). Economists and psychologists began to investigate how people value environmental change, environmental policy and ecosystem management, using stated preference and revealed preference methods (Hanley and Spash 1993). Demographers investigated the relationship between resource scarcity, population growth and migration. Decision theorists tried to develop new tools for policymaking, such as multicriteria and multiobjective decision analysis, which were fine-tuned to the often qualitative and unpriced nature of environmental goods (Nijkamp 1979). Many disciplines worked together in integrated modeling and assessment, addressing both ecosystems and global scales, recently leading to a new wave of global climate integrated assessment models. Finally, statisticians collected data and developed indicators to monitor energy use and social-economic trends in relation to the state of the environment and resource scarcity. So in the past 25 years there has been an explosion of social scientists' interest in environmental and resource issues.

It should be noted that there are many approaches to studying the relationship between the economy and the environment. This involves linking economics and ecology (Costanza *et al.* 1997). An important stream is based on materials accounting, using the principle of material balance to describe the chain of extraction, transformation, consumption and emission (Ayres *et al.* 1999). Another approach has focused on building economic and social accounting systems that can incorporate the measurement of economic welfare together with the measurement of environmental indicators. Here we will present an ecological economics approach to the study of biodiversity. This is motivated by the fact that biodiversity is a multidimensional concept linked to biological, ecological, cultural and economic entities. Our study differs from other related studies, such as Barbier *et al.* (1994), Pearce and Moran (1994), Perrings *et al.* (1995a), Rapport *et al.* (1998), and van Kooten and Bulte (2000), in the following ways: (1) it presents a stronger focus on the analysis of biodiversity rather than ecosystems or natural assets; (2) it subscribes to a multidisciplinary and integrated approach to shed light on the value of particular biodiversity elements; (3) it explores the use of applicable valuation methods and empirical studies; and (4) it analyses the role biodiversity indicators and value information can play in biodiversity policy and management. Some of these other studies provide interesting complementary information, such as on the economics of renewable resources (notably van Kooten and Bulte 2000).

1.3 Structure of the book

The sequence of chapters in the book follows the idea that analysis of biodiversity policy involves a number of steps, relating to the identification, measurement and aggregation of biodiversity values. Figure 1.1 contains an organization chart depicting the overall structure of the book, which is divided into five parts: context, bio-ecological foundations, economic foundations, economics-ecology interface, and policy and conclusions. Chapter 2 provides a definition of the notion of biodiversity, identifies different levels of life diversity, and discusses alternative perspectives on biodiversity value. The second part, Chapters 3 and 4, discusses the ecological analysis of biodiversity. Chapter 3 presents important ecological concepts and general frameworks relevant to the conceptualization of complex relationships between biodiversity and ecosystem functioning. Chapter 4 reviews ecological valuation methods, and discusses some applications. Part III, Chapters 5 and 6, discusses the economic analysis of biodiversity values. Chapter 5 discusses general aspects of the economic valuation of biodiversity benefits, and offers a classification of the different biodiversity value categories. Chapter 6 examines which valuation methods can be used for specific value types. In addition, it presents a survey of valuation studies for different levels of diversity, and critically discusses the range of empirical findings. Part IV, Chapters 7 to 9, discusses the integration of economic and ecological information to assess multidimensional biodiversity benefits. Chapter 7 explores the joint research efforts of economic and ecological sciences aimed at producing a better understanding of the complex context of biodiversity. A review is offered of frameworks and methods of integrated modeling of the relationship between biodiversity, ecosystem structure, ecosystem functions, economic activity, and human welfare. This also includes an illustration of an ecological-economic model for biodiversity evaluation. Chapter 8 discusses the use of multicriteria analyses, which can transparently aggregate economic and ecological indicators in biodiversity evaluation. Chapter 9 is devoted to meta-analytical methods, which can aggregate information obtained in previous biodiversity studies and help to 'predict' for a new biodiversity analysis. Finally, Part V, Chapters 10 and 11, concludes the book. Chapter 10 addresses the implications of the use of biodiversity value information for policy design, devoting special attention to certification and ecolabeling mechanisms. Chapter 11 presents conclusions on the roles of ecology, economics and their integration in biodiversity analysis.

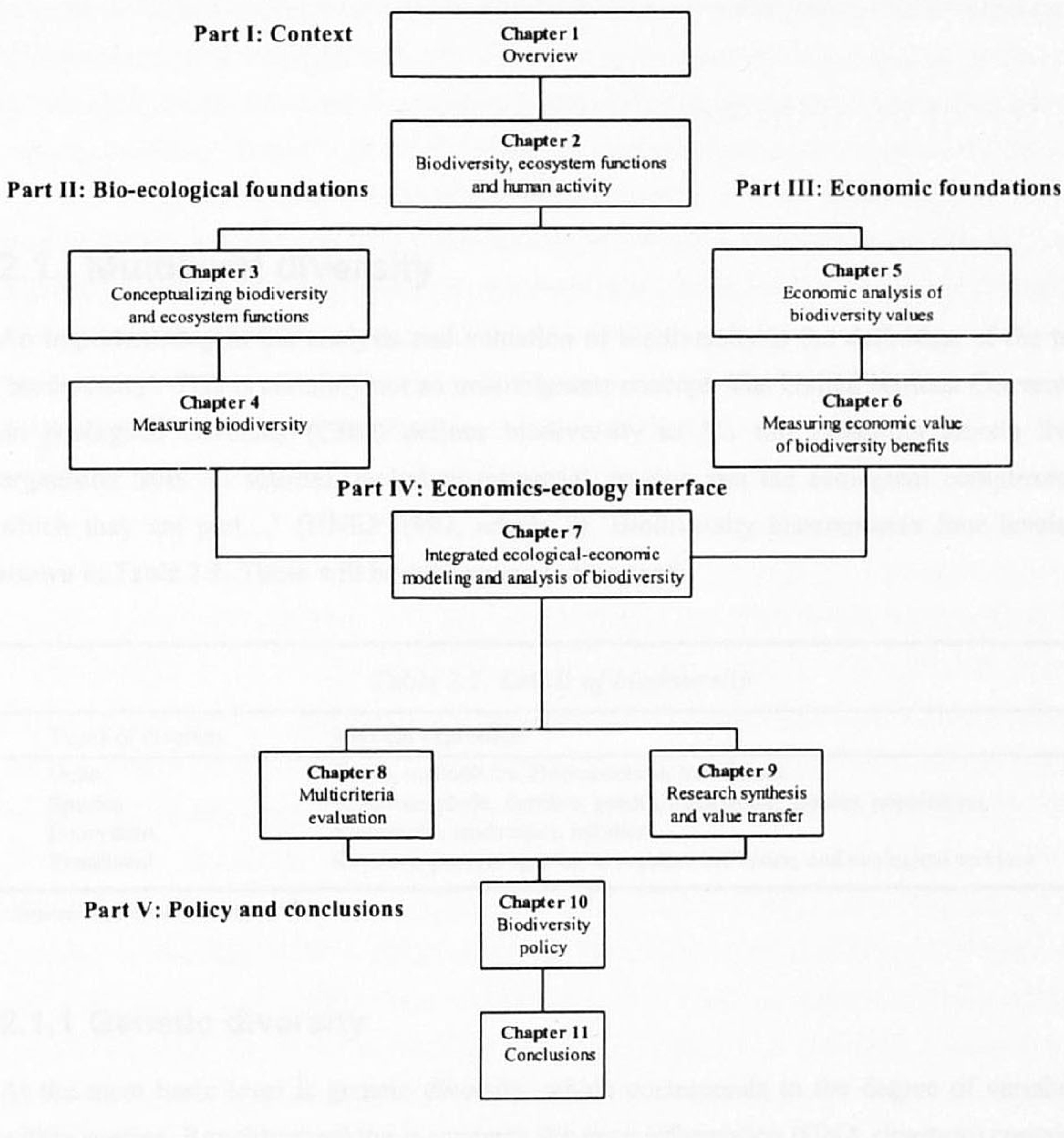

Figure 1.1: Structure of the book

The information offered in this book is not only meant to be of interest to students and teachers in environmental and resource economics, but also researchers in other fields related to ecological economics. This study aims to be a source of reference for all those concerned about the future of biodiversity.

Chapter

2

BIODIVERSITY, ECOSYSTEM FUNCTIONS AND HUMAN ACTIVITY

2.1 Multilevel diversity

An important step in the analysis and valuation of biodiversity is the definition of the term 'biodiversity'. This is certainly not an unambiguous concept. The United Nations Convention on Biological Diversity (CBD) defines biodiversity as '... the variability among living organisms from all sources, including terrestrial, marine and the ecological complexes of which they are part...' (UNEP 1992, article 2). Biodiversity encompasses four levels as shown in Table 2.1. These will be subsequently discussed.

Table 2.1: Levels of biodiversity

Types of diversity	Physical expression
Gene	Genes, nucleotides, chromosomes, individuals
Species	Kingdom, phyla, families, genera, subspecies, species, populations
Ecosystem	Bioregions, landscapes, habitats
Functional	Keystone process species, ecosystem resilience, and ecological services

Source: Turner *et al.* (1999).

2.1.1 Genetic diversity

At the most basic level is genetic diversity, which corresponds to the degree of variability within species. Roughly speaking it concerns the gene information (DNA structure) contained in the genes of individual plants and animals. Much of the current debate over biodiversity relates to the extinction of the genetic pool available for future generations and the associated implications for still undeveloped medicines, crops, pharmaceuticals products, petroleum substitutes, and other products and amenities that may never come to light (Wilson 1988b).

Gene diversity provides the basis for bio-technological manipulation of genetic material. In the domain of agriculture, for example, the manipulation of genetic material has been proven to be an issue of major economic importance. It affects comparative advantage in international trade and, as a result, has played an important role in international commercial agreements such as the GATT/WTO. The global debate on the sources underlying the loss of genetic information has been focused on the relation between the providers of genetic diversity, primarily the developing world (e.g., tropical rain forest countries), and the beneficiaries of the genetic diversity, predominantly developed countries. In this context, the Convention on Biological Diversity seeks to give '... priority access to a fair and equitable basis by contracting parties, especially developing countries, to the results and benefits arising from biotechnology based upon genetic resources provided by those contracting parties...' (UNEP 1992, article 19.2).

In short, the extent to which biodiversity is lost depends on the extent to which genetic diversity is lost which, in turn, depends to the extent to which gene information is lost. The value assessment of such information loss can be based on the conceptualization of 'genetic difference' and 'genetic distance' within species (Weitzman 1992; 1993; 1995).

In order to describe the diversity between species it is necessary to go one step further in the level of organization of living resources. We refer to species diversity and this will be discussed in the following section.

2.1.2 Species diversity

Species diversity refers to the variety of species on earth, or in a given region. This is associated with a large degree of uncertainty. In fact, estimates of the total number of species on earth range from 5 - 300 million, of which about 1.5 million have been described, and less than 0.5 million have been analyzed for potential economic benefit properties (Miller *et al.* 1985; CBD 2001). The best-catalogued species groups include vertebrates and flowering plants, with other groups, such as lichens, bacteria, fungi and roundworms, relatively under-researched (Wilson 1988a; Pimm *et al.* 1995). Such a lack of information has important implications for defining priorities for cost-effective conservation. In the domain of the natural sciences, scientists have long been proposing the concept of 'biodiversity index' as an important tool in conservation policy. This index is devoted to summarizing information about the number of different species within a given area, the implicit survival probability distribution functions, and the degree of variety in the relationships of species to each other (Margalef 1996). Without such a tool, conservation of the incorrect range of species can be encouraged. For example, according to the results of an empirical study conducted by Solow *et al.* (1993), minimizing the probability of crane species loss does not always correspond to minimizing the loss of variability among living crane species. This reasoning implies that the US Endangered Species Act, which sets its priorities in such a way that whatever is the most threatened should be conserved first, may not minimize the expected level of diversity loss. Weitzman (1998) presents a theoretical argument closely related to this idea, arguing that if what is valued by society is maximum species diversity, then preservation efforts should be focused on the threatened species that are genetically most distant from other species. He shows that optimal conservation of biodiversity is impossible without a sense of the magnitude of the appropriate species 'distinctiveness', extinction probabilities, and the costs of improving species survival.

Saving species solely according to the degree of threat of extinction tends to ignore the reason why the resource is severely threatened in the first place. If the cause of extinction is not very amenable to policy measures, allocating resources towards conservation is likely to be wasteful anyway (Pearce 1999). More recently, van der Heide *et al.* (2002) have also argued that Weitzman's proposal may lead to undesirable policies, because it neglects ecological relationships, focusing entirely on genetic distances.

2.1.3 Ecosystem diversity

Ecosystem diversity refers to diversity at the community level, i.e., at a supra-species level. This covers the spatial variety of ecosystem types, which include living communities of organisms, their respective natural habitats and the physical conditions under which they live. A long-standing theoretical paradigm has predicted that species diversity is important because it enhances the productivity and stability of ecosystems (Odum 1950). Recent studies, however, acknowledge that no pattern or determinate relationship needs to exist between species diversity and the stability of ecosystems (Johnson *et al.* 1996). Folke *et al.* (1996) instead suggest that ecosystem diversity may be linked to the prevalence of a limited number of organisms and groups of organisms that seem to drive or control the critical processes necessary for ecosystem functioning – known as 'keystone processes'. The disappearance of these processes reduces the ecosystem's capacity to accommodate external shocks, like climatic and human influences, and ultimately results in the loss of spatial variety in ecosystem types.

Closely related to the notion of loss of keystone processes are the notions of ecosystem stability and resilience. The latter has been defined in ecology as an extended stability concept. It comes in two variants. One refers to the properties of the ecosystem near some stable equilibrium. This definition, attributed to Pimm (1984), is concerned with the time it takes for a disturbed system to return to some initial state, i.e., the resilience of an ecosystem is measured by its speed of return to equilibrium. The second variant is concerned with the magnitude of disturbance that can be absorbed before an ecosystem is displaced from one state to another. This definition, attributed to Holling (1973; 1986; and 1992), is concerned with the ability of an ecosystem to maintain its self-organization without undergoing the destructive and possibly irreversible change involved in crossing the threshold between stability domains. Therefore, analyzing ecosystem resilience is about determining the limits within which the different state variables can be disturbed without flipping the current ecosystem to another regime of behavior (Holling *et al.* 1995; Reggiani *et al.* 2002).

Ecosystems are characterized by a hierarchical structure, where each level comprises a different temporal and spatial scale (Gibson *et al.* 2000) – and for this reason most ecological models tend to be highly disaggregated. Not only is the resilience of a system different at different ecosystem hierarchy levels, but also in each state and at each level it depends upon its ability to cycle through different states at another level (Perrings 1998).

For any ecosystem structure to be sustained, a minimum level of variety of communities of living organisms and their abiotic environments is required. Unfortunately, quantitative indicators of ecosystem diversity are neither directly available nor easy to measure. As a consequence, ecosystem robustness is still little understood. Often the resilience thresholds for ecosystem keystone processes associated with different environmental conditions at different temporal and spatial scales are not known (Perrings and Pearce 1994).

2.1.4 Functional diversity

According to Turner *et al.* (1999), functional diversity refers to the variety of ecosystem functions. It is the outcome of the interactions between the structure and processes of ecosystems. Ecosystem structure refers to the tangible items such as plants, animals, soil, air and water of which an ecosystem is composed. Ecosystem processes refer to the dynamics of transformation of matter or energy between living and abiotic systems. These can involve interactions between hydrological and geomorphological systems, ecosystem fauna and flora, and photosynthesis and food web support. These processes are subsequently responsible for the provision of services – life support services – such as the assimilation of pollutants, cycling of nutrients, soil generation and preservation, pollination of crops, and maintenance of the balance of gases in the air (Maltby *et al.* 1996a and 1996b). They also enable the development and maintenance of an ecosystem structure that is, in turn, the basis for the continued provision of goods and services. Ecosystem functions are the result of interactions between the structure and its processes.

The task of evaluating the structure and functioning of an ecosystem requires that much be known about what the ecosystem does and what that is worth for both biodiversity and for humans. The value of ecosystem structure is generally more easily appreciated than that of ecosystem functioning. Assessing ecosystem functions, such as nutrient retention and pollution absorption for any given region, is extremely difficult. But ecosystem structure is also incompletely known. To assess the value of, for instance, the insect fauna and soil fungi, when many of these species have never even been described taxonomically, pushes human knowledge beyond its current limits (Westman 1985).

The preservation of ecosystem processes and their consequent functioning is as important a goal for conservation as is the preservation of ecosystem structure. Ecology has now come to understand ecosystem processes to the extent that some management principles are evident, even if many questions remain unsolved.

2.2 Loss of biodiversity

The current occurrence of biodiversity loss is a consequence of the decisions of billions of individual users of biodiversity products and service flows. This is a result of the 'unpriced scarcity' and 'lack of property rights' nature of the environment, biological resources, and biodiversity in particular. The notion of 'externalities' is relevant here. It means that the social value of various biodiversity goods and service flows is not or is insufficiently reflected in market prices. As a result, a socially undesirable level of provision of these goods and services will result – in economic terms, there is no Pareto-optimal situation. Second, many human activities have contributed to an unprecedented rate of biodiversity loss, which threatens the stability and continuity of ecosystems as well as their provision of goods and services to mankind (Pimm *et al.* 1995; Simon and Wildavsky 1995). One can question why, when biodiversity generates so many benefits for humans, it has been ignored and biodiversity loss has been allowed to occur. The answer is that many biodiversity services are public goods, available for consumption and usage at no market cost, which leads to their destruction.

When thinking about the driving forces behind the loss of biodiversity, it is current practice to distinguish between 'proximate' and 'fundamental' causes of biodiversity loss. According to Pearce and Moran (1994, p. 18) proximate causes '... show up as the more popular explanations of biodiversity loss...' whereas fundamental causes '... lie behind the proximate causes and are rooted in economic, institutional and social factors...'

Proximate causes include the exploitation of species, environmental pollution, and the degradation of natural resources (Perrings *et al.* 1995b). Ecologists estimate that less than one-tenth of 1 per cent of naturally occurring species are directly exploited by humans (Wilson 1988a; CBD 2001). It is therefore argued that the major threat to (species) biodiversity loss is not caused by direct human exploitation of species, but by habitat changes and the degradation that results from the expansion of human population and human activities (Ehrlich 1988; McNeely *et al.* 1995). Furthermore, the human pressure on biodiversity is now beginning to pose substantial risks to the stability of a variety of regional ecosystems, creating threats to the economic and cultural lives of many societies.

Biodiversity loss also links local and global problems (see Swanson *et al.* 1997). The importance of the spatial element arises from a reciprocal relationship: (1) local processes have global impacts; and (2) global trends give rise to local effects. Table 2.2 shows historic conversion rates of natural habitats to agriculture. Such pressure, depending not only on a spatial setting but also on the socio-economic, political and cultural setting, will affect the structure and functions of ecosystems, resulting in impacts on global climatological conditions and geochemical cycles; in the modification of carbon sink functioning with global warming consequences; and in soil erosion, downstream sedimentation, flooding and salinization.

While the proximate causes of biodiversity loss relate to the worldwide trend of human population growth, and its impact on production and consumption patterns, the fundamental causes of biodiversity loss are associated with the conditions within which biological resource use and land use decisions are made. Two important fundamental causes underlie the loss of biodiversity. The first relates to market failures and the second to the lack of property rights. Many biodiversity benefits, such as the ability to supply clean air, are not 'cashed' flows, i.e., there is no market price mechanism that fully captures such benefits. In other words, markets fail to internalize biodiversity protection benefits. These are external effects, i.e., unintended effects outside the market on the welfare or productivity of other humans. In such a context, the rate of return on biodiversity conservation will almost certainly fail to compete with the rate of return on development projects. Turner and Jones (1991) refer to this as 'interrelated market and intervention failures'. These, in turn, create a fundamental failure of information, i.e., a lack of understanding of the multitude of values that may be associated with the conservation of biodiversity.

The lack of property rights is regarded as an even more basic cause of the loss of biodiversity (Clark 1990; Hardin 1993). A classic example is water pollution from agriculture. The supply of fresh water, an essential feature of many ecosystems, is a public good and is exposed to 'open access' farming pressure. A lack of enforceable property rights allows the unrestricted depletion of the resource and causes negative externalities to society. But even when land is privately owned, many ecosystem functions provide off-site benefits that the resource's owner is unable to appropriate. The lack of a market for these off-site

Table 2.2: Conversion of natural habitat to agriculture

Regions	1900 cropland (millions of ha)	1980 cropland (millions of ha)	Change (%)
Sub Saharan Africa	73	222	+ 204
Latin America	33	142	+ 330
South Asia	89	210	+ 136
China	89	134	+ 51
South-East Asia	15	55	+ 267
North America	133	203	+ 53
Europe	145	137	– 5
(ex) USSR	147	233	+ 58

Source: Pearce and Moran (1994).

benefits limits the incentive to keep land free from agricultural development, since the private benefits derived by the owner do not reflect the full benefit to society.

2.3 Alternative perspectives on biodiversity value

Given the four levels of life diversity and the multiplicity of benefits, it should be evident that there is no single notion of biodiversity value. This section presents additional considerations that support this belief (see Nunes and van den Bergh 2001).

2.3.1 Instrumental vs. intrinsic values

Many people do not feel comfortable with placing an instrumental value on biodiversity. The common argument is that biodiversity has a value on its own – also known as 'intrinsic value'. A more extreme version of this argument claims that instrumental monetary valuation is a nonsense exercise (Ehrenfeld 1988). Many others, however, accept the placement of a monetary value on biodiversity, arguing that this merely makes explicit the fact that biodiversity is used for instrumental purposes in terms of production and consumption opportunities (Fromm 2000). This is based on the idea that making public or private decisions that affect biodiversity implicitly means attaching a value to it. Finally, monetary valuation can be considered a democratic approach to deciding about public issues, including those affecting biodiversity conservation or loss.

2.3.2 Monetary vs. biological indicators

The monetary valuation of biodiversity is anchored in an economic perspective (see Randall 1988). Such an evaluation is essentially based on market transactions and conceives biodiversity as a tradable good with human-based value. Monetary indicators can serve as a common unit for the comparison and ranking of alternative biodiversity management policies.

In contrast, biological assessments of biodiversity value give rise to non-monetary indicators. These include, for example, species and ecosystems richness indices (see Whittaker 1960 and 1972), which have served as important valuation tools in the definition of 'Red Data Books' and 'Sites of Special Scientific Interest' (see Chapter 4). It is not guaranteed, however, that monetary and biological indicators will point in the same direction. They should, at best, be regarded as complementary methods for the assessment of biodiversity changes. Nevertheless, economic indicators should, whenever possible, indirectly use accurate biological indicators.

2.3.3 Direct vs. indirect values

The notion of direct value of biodiversity is sometimes used to refer to human uses of biodiversity in terms of production and consumption. The notion of indirect value of biodiversity has been associated with a minimum level of ecosystem infrastructure, without which the goods and services provided by it would cease to exist (Farnworth *et al.* 1981). Barbier (1994, p. 156) recently described the 'indirect value' of biodiversity as '... support and protection provided to economic activity by regulatory environmental services...'. In the literature one can find other terms, such as the 'contributory value', 'primary value', and 'infrastructure value' of biodiversity, all of which seem to point at the same notion (see Norton 1986; Gren *et al.* 1994; and Costanza *et al.* 1998). Some of these authors subscribe to the opinion that measurement of biodiversity benefits is possible, but that it will always lead to an underestimation of their 'real' value, since the 'primary value' of biodiversity cannot be translated into monetary terms. As Gowdy (1997) has recently remarked, '... although values of environmental services may be used to justify biodiversity protection measures, it must be stressed that value constitutes a small portion of the total biodiversity value...'.

2.3.4 Biodiversity vs. biological resources

The term 'biodiversity' has often gone hand in hand with the term 'biological resource'. These terms, however, do not reflect the same notion. Whereas biodiversity refers to the variety of life, at various levels, biological resources refer to the manifestation of that variety. According to David Pearce (1999, p. 2) '... much of the literature on the economic valuation of "biodiversity" is actually about the value of biological resources and it is linked only tenuously to the value of diversity...'. The precise distinction is not always clear, and the two categories seem to be somewhat overlapping. Therefore, care is needed in interpreting studies that claim to present economic values of biodiversity. We will mainly use here the UNEP (1992) definition, as contained in Table 2.1.

2.3.5 Levels of vs. changes in biodiversity

Economists stress that valuation should focus on changes rather than on levels of biodiversity. Non-economists have, however, fallen into the trap of valuing levels. For instance, we may refer to the value assessment of ecosystem services and natural capital for the entire biosphere (Costanza *et al.* 1998). Economic-theoretical support for such a valuation approach is usually weak and incomplete. One reason is that willingness to pay and willingness to accept are welfare measurements based on compensation or equivalence monetary variations relating to a change (see Chapter 5). These changes should be small in comparison with an individual's income level.

2.3.6 Local vs. global diversity

The design of a valuation context involves important decisions about the spatial frame of analysis (Norton and Ulanowicz 1992). Whereas biodiversity loss is usually considered in a global or worldwide context, biodiversity valuation studies frequently address policy changes or scenarios defined at local, regional or national levels. Although this seems contradictory, biodiversity and its loss are actually relevant at multiple spatial levels, ranging from local to global (Hammond *et al.* 1995). It is clear that due attention for the geographical scale of biodiversity is a sine qua non for comparative study.

2.3.7 Genetic vs. other life organization levels

Scientists face an important decision when valuing biodiversity, namely, which level of diversity to consider. Some scientists, especially those from the natural sciences, tend to focus on genetic and species levels, whereas others, notably social scientists, tend to study biodiversity at the level of species and ecosystems. This is of course caused by different frames of reference; social scientists tend to adopt a more systematic approach as part of their valuation and assessment research. Unresolved issues are whether studying biodiversity at multiple levels leads to double counting, and whether sufficient information is available at each biodiversity level necessary to perform valuation studies. The lesson is that ecological and social disciplines have to work together much more on concrete real-world research issues.

2.3.8 Holistic vs. reductionist approaches

According to the holistic perspective, biodiversity is an abstract notion, linked to the integrity, stability and resilience of complex systems, and thus is difficult to disentangle and measure (Faber *et al.* 1996). In addition, there is insufficient knowledge and understanding of the human and economic significance of almost every form of life diversity, further complicating the translation of physical-biological indicators of biodiversity into monetary values. For these reasons, the economic valuation of biodiversity is regarded by many scientists as a hopeless task (Ehrenfeld 1988). In contrast, the reductionist perspective is based on the idea that one is able to disentangle, or disaggregate, the total value of biodiversity into different economic value categories, notably direct use and passive use or nonuse values (Pearce and Moran 1994). Clearly, still more methodological challenges have to be resolved here.

2.3.9 Expert vs. general public assessments

Monetary valuation as a basis for public policy relies on the premise that individuals with varying educational levels and life experiences can participate in the valuation of biodiversity changes. Another view assumes that laypersons cannot judge the relevance and complexity of biodiversity-ecosystems-functions relationships. Instead, judgments and valuation of biodiversity changes should be left to experts, notably biologists. An example of an intermediate 'solution' is to let experts inform laypersons sufficiently before they are confronted with a valuation exercise (NOAA 1993). There is certainly a need for more applied field experiments.

It is clear from the above concise overview that many different biodiversity value perspectives can be distinguished based on various combinations of the above nine considerations. Evidently, it is crucial to know the perspective adopted in a particular study before comparing it with others. This requires a careful documentation of hypothesis and research protocols in empirical studies, and this is only beginning to materialize.

PART II
Bio-Ecological Foundations

Chapter

3

CONCEPTUALIZING BIODIVERSITY AND ECOSYSTEM FUNCTIONS

3.1 Introduction

Biodiversity is a multifaced concept with both ecocentric and anthropocentric characteristics showing a great variety in all regions of our world. The analysis of biodiversity is, consequently, rooted in the domain of both the natural and the social sciences. Its modeling implies knowledge of the relationships between biodiversity, the dynamics of ecosystems, and the level of human economic activities. One reason why biodiversity modeling has been so difficult is the complex and partly unobservable character of these relationships, as illustrated in the example of a terrestrial ecosystem in Figure 3.1. The geographic diversification in biodiversity and its interrelatedness to socio-economic and physical-climatological conditions makes it also difficult to develop and apply an operational methodology for biodiversity analysis. This strand of research is certainly still in its infancy.

Essentially, the biological organization base of an ecosystem is made up of three main interrelated classes: (1) biotic resources emerging from the soil or water (such as vegetation and animal populations); (2) abiotic resources with a productive or consumptive nature (such as minerals and energy); and (3) environmental components needed for human well-being (such as clean water or fresh air). In general terms, modeling such biological organization has emerged from two streams of ecology, namely community and ecosystem ecology (Holling 1992; Holling *et al.* 1995; and Schindler 1990, 1995). Community ecology emphasizes the study of the interrelations between species. Some applied studies have, however, called attention to situations where the abiotic environment plays an important role in (re)shaping the relationships of the ecological community. These situations offered a new impetus to ecological thought, giving birth to a second stream in ecology literature: ecosystem ecology.

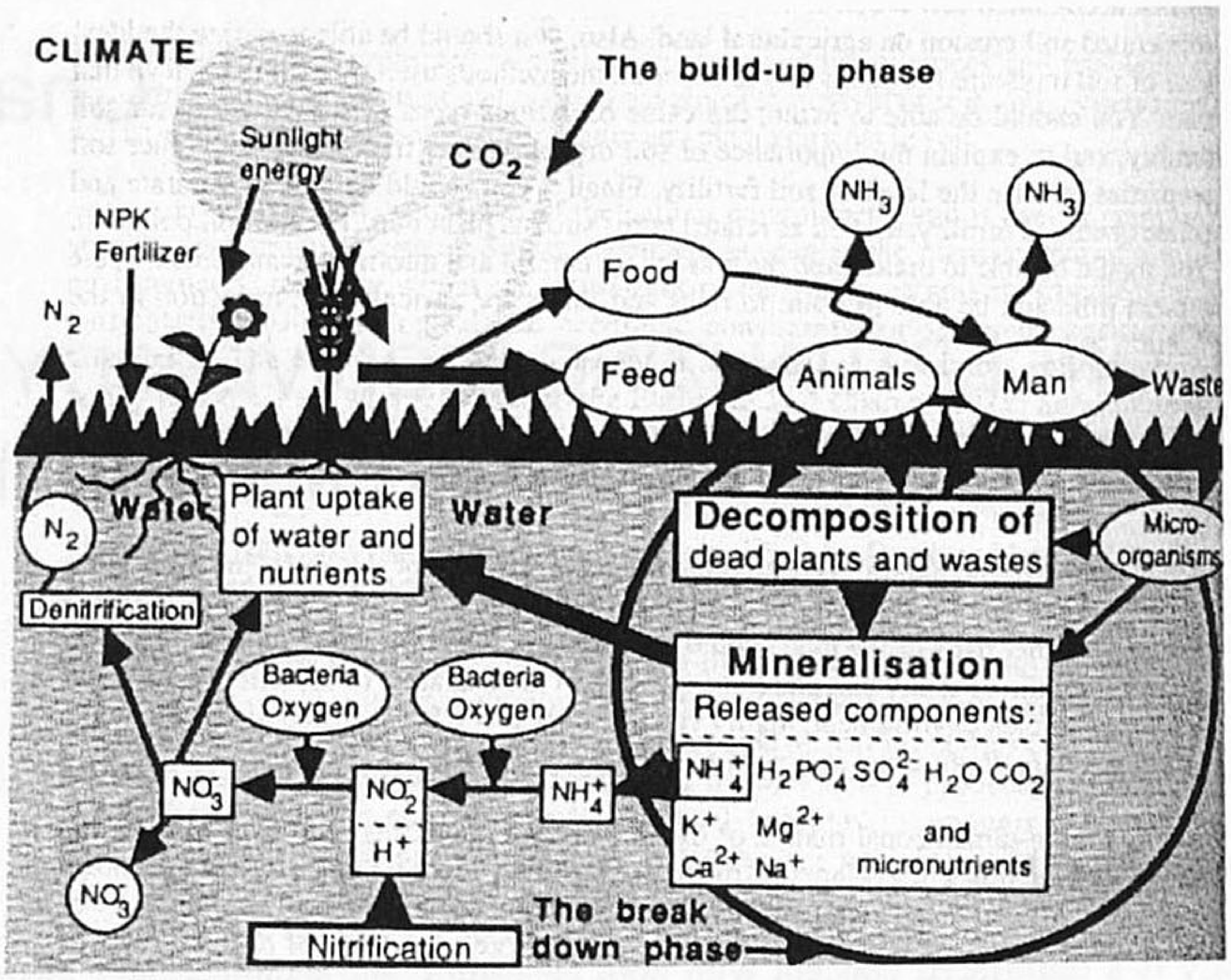

Figure 3.1: The dynamics of a terrestrial ecosystem
Source: UNEP (1995).

Ecosystem ecology takes biotic and abiotic elements as variable and interactive. For instance, abiotic diversity (e.g., physical characteristics of the landscape such as soil pH and salinity) is expected to be linked to the prevalence of endemic species and thus to biotic diversity and rarity in a natural way (Bertollo 1998).

In this context, ecological valuation methods are not only aimed at assessing diversity and rarity of species, but also at assessing the complex interactions between the biotic and abiotic environments, based on the assumption that the variety of abiotic conditions is equally important as the variety of species. In other words, ecosystem ecology aims to identify and characterize the impact of biotic-abiotic interactions on food webs and species interrelations and to assess the role of nutrient flows. Independent of the ecological modeling approach, an important aspect of ecosystem ecology is the recognition that the variability of the biological resources influences the functioning and structure of ecosystems. Without attempting to be exhaustive, we outline four main approaches to modeling in the following sections.

3.2 The '4-box model'

Holling (1987 and 1992) proposed a model to describe and explain the dynamics of a terrestrial ecosystem. Its structure is characterized by a sequence, or, more precisely, cycle, of four phases. These are: (1) exploitation; (2) conservation; (3) release, and (4) reorganization (see Figure 3.2).

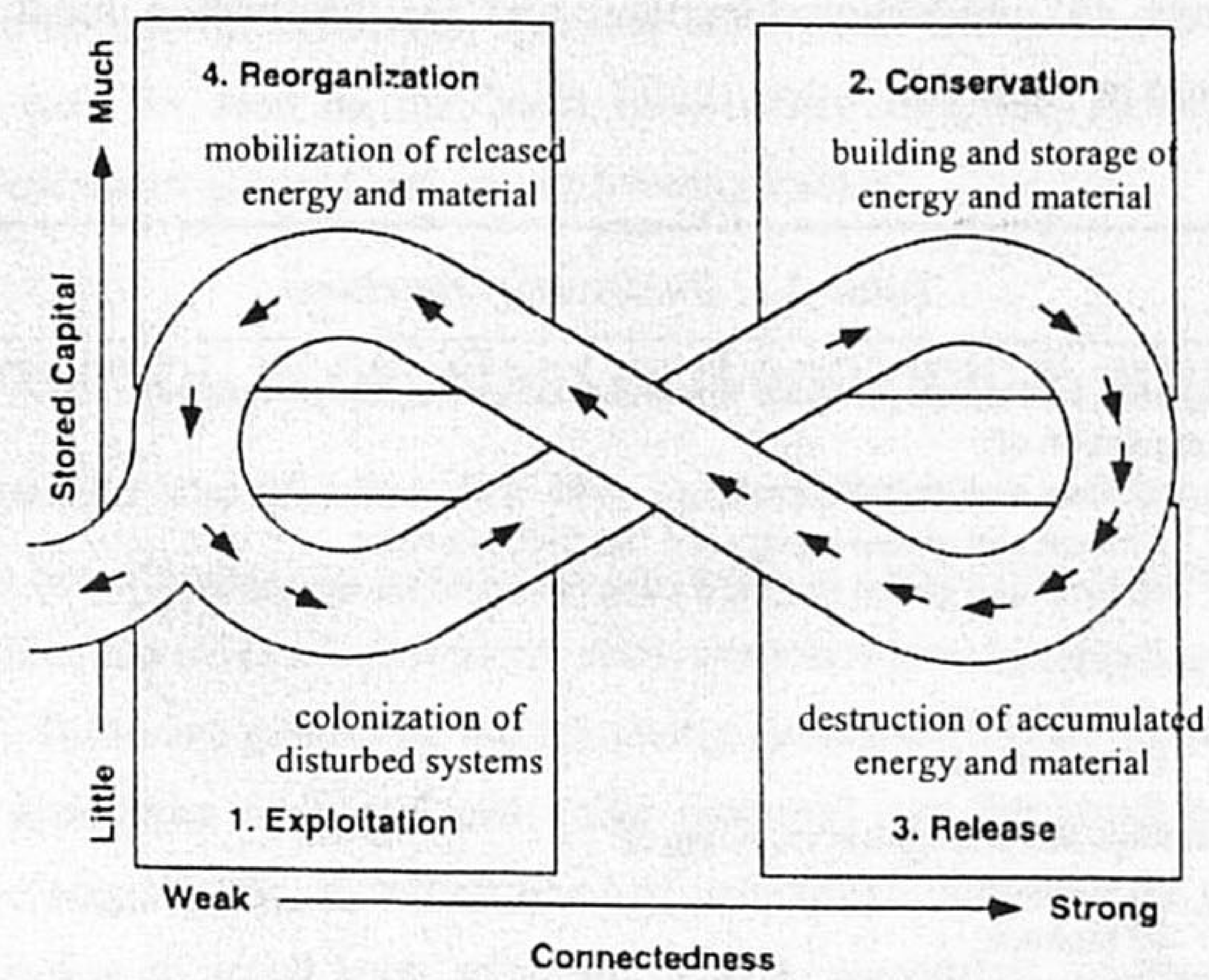

Figure 3.2: The structure and functions of an ecosystem

Source: Holling (1992).

Within this model, ecosystems develop from the exploitation phase, during which systems easily capture accessible resources, to the conservation phase, during which they build increasingly complex structures. The exploitation function refers to the ecosystem's processes, which are responsible for 'colonizing disturbed sites'. The exploitation phase in ecology can be compared to pioneer human societies colonizing new territories. Ultimately they shift to the conservation phase, in which systems build and store energy and material. Botanical gardens and arboreta constitute examples of ecosystems at the conservation phase. The system is then in a condition to make some of the mature structures and available nutrients free in the release phase. Examples of factors initiating release are fire, storms, and pests (Costanza *et al.* 1995). This phase is followed by the reorganization phase. This refers to the ecosystem processes that are responsible for mobilizing released energy and materials and making them available for the next exploitation phase. This development has given much support to the quantitative analysis of biodiversity, although it ought to be recognized that the development toward statistically testable modeling approaches is still weak.

3.3 Biodiversity functions

Biodiversity research is often based on a functional typology. Based on a categorization of ecosystem functions initially developed by Odum (1971) and de Groot (1994) one can distinguish between four main categories of biodiversity functions: (1) life support functions; (2) carrier functions; (3) production functions; and (4) information functions. These are further disaggregated in Table 3.1.

Table 3.1: Biodiversity functions

Life Support Functions (output: fundamental ecological processes)

a) regulation of:
- i) local and global climate;
- ii) local and global energy and nutrient balances;
- iii) local and global chemical composition of the air and water.

b) protection of:
- i) soil erosion;
- ii) watershed.

c) storage and absorption/recycling of:
- i) nutrients;
- ii) biomass;
- iii) human waste.

Carrier Functions (output: provision of space)

a) human habitation;
b) agriculture;
c) recreation;
d) nature protection.

Production Functions (output: provision of environmental resources)

a) oxygen;
b) water (for drinking, irrigation, industry);
c) food and nutrients;
d) fuel and energy;
e) raw materials (for building, construction and industrial use);
f) genetic resources (for medicinal and pharmaceutical use).

Information Functions (output: provision of opportunities for human contemplation)

a) spiritual, religious or moral inspiration;
b) aesthetic experience;
c) historic and educational information;
d) cultural and artistic inspiration.

Source: de Groot (1994).

Biodiversity is seen to have a life support function, namely the regulation of essential ecological processes. Life support functions include the maintenance of a healthy environment, and the provision of clean air, water and soil, as well as flood control, carbon storage and waste absorption. Most life support functions are fuzzy, and cannot be easily demarcated and identified. The carrier functions refer to the provision of space for human activities such as habitation, agriculture, and recreation. The production functions refer to the provision of environmental resources, ranging from raw materials for industrial use to water and energy resources. The information functions refer to the maintenance of mental health, providing opportunities for reflection, spiritual enrichment and aesthetic experience. These four functions may be seen as the basic biodiversity functions, which represent in an aggregate form all major components of biodiversity values.

3.4 Community, ecosystem and ecological organization

The above presented classification has also generated critical reflection. More recently, Norberg (1999) has proposed an alternative approach to classifying ecosystem functions and services of nature. He selected groups of ecosystem services to which common ecological concepts apply. These are guided by the following questions: are the goods and the services internal to the ecosystem or shared with other systems? Are the goods and the services of biotic or abiotic origin? And, at which level of ecological hierarchy are goods and services maintained? Bearing in mind such selection criteria, ecosystem functions and services of nature are classified into three categories: (1) maintenance of the populations; (2) regulation of material and energy flows; and (3) organization of biological units through selective processes (see Table 3.2). These categories relate to three major fields in ecology that have well-established theoretical foundations, namely population/community ecology, ecosystem ecology, and biological organization research, respectively (Levin *et al.* 1997; Levin 1998).

The first category corresponds to the group of ecosystem services that are '… associated with certain species or a group of similar species…' (Norberg 1999, p. 185). Examples of such services include valuable food products and goods such as fish, timber, pharmaceuticals, and flowers. The second category consists of processes that regulate exogenous chemical or physical cycles, i.e., processes that drive material and energy flows within ecosystems. Important categories include global cycles of chemical compounds such as water, CO_2, nitrogen, sulphur and phosphorus. The third category of ecosystem services is related to the organization of biotic entities. Organization is present at virtually all scales: organization of genes through natural selection, spatial distribution of a population through dispersal and competitive exclusion, and food webs and ecosystems through invasion and extinction

Table 3.2: Community, ecosystem and organization hierarchy

Maintenance of Populations (output: the goods and the services are internal to the ecosystem)

a) fish;
b) timber;
c) pharmaceuticals;
d) flowers.

Regulation of Material and Energy Flows (output: chemical and physical cycles across ecosystems)

a) water;
b) nitrogen;
c) CO_2.

Organization (output: organization of the biotic entities)

a) organization of genes;
b) spatial distribution of a population;
c) food webs and ecosystems.

Source: Norberg (1999).

processes. Some of the research analyzing these processes occurs on the boundary of ecology and evolutionary biology. The latter, moreover, provides a full explanation of the dynamics of biodiversity, including its innovation and increase. It goes without saying that much more fundamental and applied research is needed in this field.

3.5 Ecosystem health

The definition of 'ecosystem health' indicates how well an ecosystem is functioning relative to its potential performance. In addition, it reflects how important it is for the functioning of other ecosystems and, ultimately, for the functioning of the biosphere (Sijtsma *et al.* 1998). In other words, well-functioning ecosystems are considered to be more 'healthy' than malfunctioning systems. From an ecological perspective, an ecosystem is said to be healthy if it is stable and sustainable, that is, if it is active and maintains its organization and vigor over time and is resilient to stress (Costanza 1992).

A health index is an overall measure of ecosystem integrity that takes into account both ecological and human processes. The measurement of ecological processes involves the identification of biotic and abiotic parameters or indicators, which represent soil, flora and fauna conditions. It also requires the definition of a spatial frame or the demarcation of a relevant geographical area (Norton and Ulanowicz 1992; van den Bergh *et al.* 1996). Commonly, spatial boundaries are drawn in accordance with the ecosystem's geophysical characteristics (e.g., wetland, marine, and terrestrial ecosystems). When spatial boundaries are clearly defined, important questions regarding the structure of the ecosystem can still remain unanswered whenever the ecosystem is originally bounded and is described as a whole. There is a need to describe the ecosystem in terms of diverse parts, making the use of spatial boundaries at a smaller scale. Furthermore, the set of biotic and abiotic indicators can differ substantially from part to part, that is, from ecosystem to ecosystem, and with the respective health index being individually assessed (Costanza *et al.* 1992). Thus, the identification of a proper geographical region and a relevant time horizon for biodiversity research is an issue fraught with many methodological pitfalls and uncertainties. After having decided upon the biotic and abiotic indicators, the time scale, hierarchy of analysis, and the targeted economic groups, the scientist is in a good position to proceed with the measurement of the overall ecosystem performance, i.e., ecosystem health – see Figure 3.3.

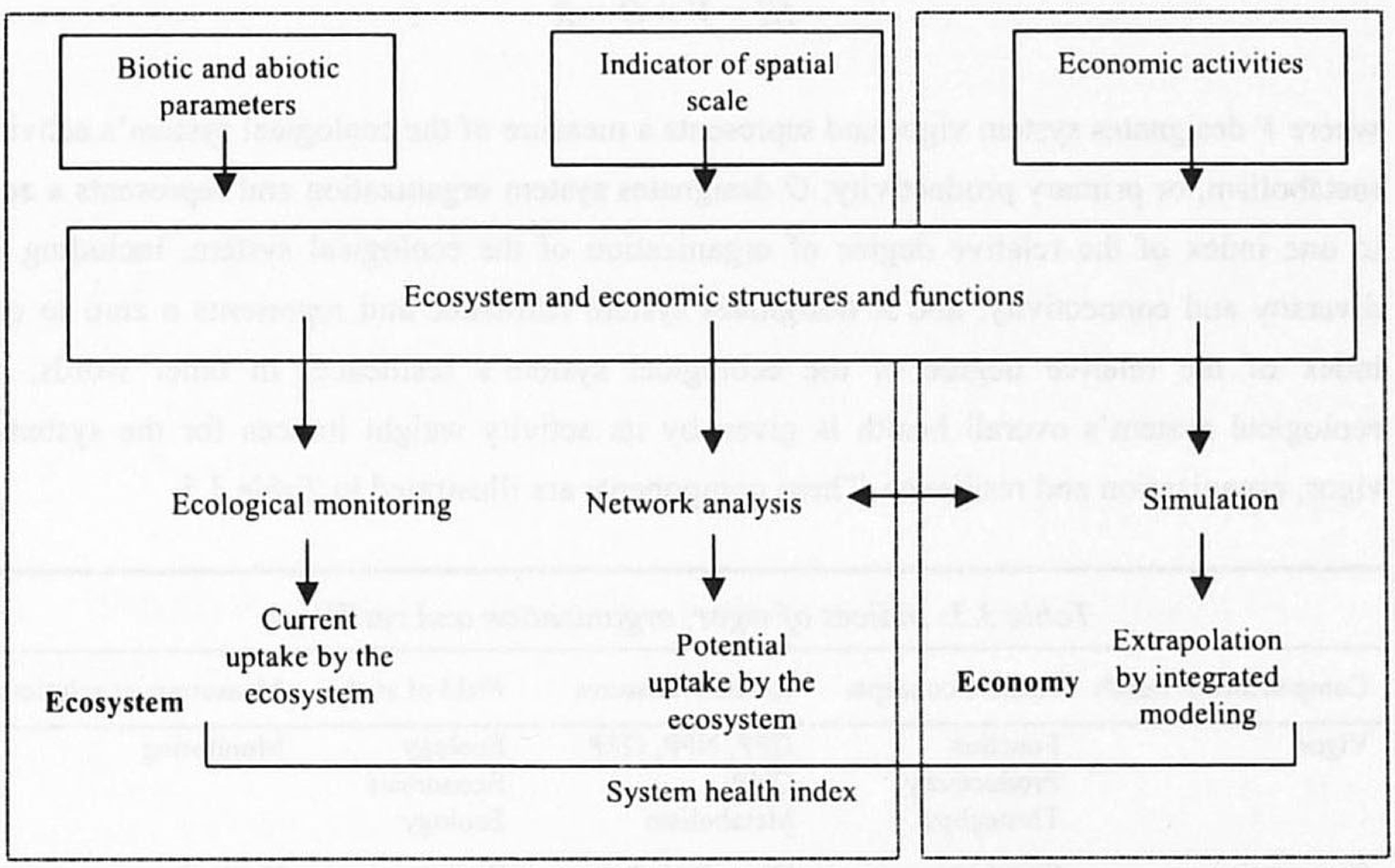

Figure 3.3: Definition of ecosystem health
Source: Nunes *et al.* (2001).

One possible way to proceed is to use directly the data derived from monitoring activities of the ecosystem, ecological structures and processes. This data will provide information about the ecosystem's real uptake, allowing us to measure overall ecosystem health. Whenever monitoring is difficult, one can integrate the available data to develop an analytical ecological framework by exploring the use of network analysis. At this stage, the scientist is not only able to measure the current ecosystem health index, but also to explore the dynamics of ecological modeling in order to estimate the potential uptake by the ecosystem. Finally, the scientist is able to simulate different scenarios by manipulating the observed human processes and indicators. For example, one can compute diverse ecosystem health indexes for alternative human impacts, or, for example, according to alternative ecosystem policy management scenarios.

The construction of ecosystem health indexes, along with the associated simulation modeling of alternative scenarios, allows policymakers to predict an ecosystem's response to various site-specific management alternatives and natural changes. Such information will provide crucial guidance for evaluating and ranking alternative ecosystem management policies. The following general formulation of an overall system Health Index *(HI)* has been proposed by Costanza (1992):

$$HI = V \times O \times R$$

where V designates system vigor and represents a measure of the ecological system's activity, metabolism, or primary productivity; O designates system organization and represents a zero to one index of the relative degree of organization of the ecological system, including its diversity and connectivity; and R designates system resilience and represents a zero to one index of the relative degree of the ecological system's resilience. In other words, an ecological system's overall health is given by its activity weight indices for the system's vigor, organization and resilience. These components are illustrated in *Table 3.3*.

Table 3.3: Indices of vigor, organization and resilience

Components of health	Related concepts	Related measures	Field of study	Measurement solution
Vigor	Function	GPP, NPP, GEP	Ecology	Monitoring
	Productivity	GNP	Economics	
	Throughput	Metabolism	Ecology	
Organization	Structure	Diversity index	Ecology	Network analysis
	Biodiversity	Mutual information predictability	Ecology	
Resilience	Integrity	Scope for growth	Ecology	Simulation modeling
		Policy scenarios	Economics	

Source: Costanza (1992).

To operationalize the vigor, organization and resilience components, the health index requires the application of different methods of measurement, involving the use of expertise from both the natural and social sciences. One example of an ecosystem health index, Ulanowicz's ascendancy index, allows for the integrated, quantitative, hierarchical measurement of ecosystem health (Ulanowicz 1992). This index measures any degradation of the system. However, it requires data on all transfers occurring in the ecosystem under consideration. The collection of this data is usually a laborious and expensive task. For this reason, fully quantified networks of ecosystems still remain scarce (Costanza 1992, Heywood 1995, Rapport *et al.* 1998).

In conclusion, biodiversity research is based on a blend of natural and social sciences approaches. In the past decade substantial progress has been made in this domain, but still many fields need to be developed further, in particular the definition of relevant indicators as a proper space-time scale and the development of statistically satisfactory, applied models.

[illegible] The collection of this data provides quality assessments and extensive [illegible] [illegible] [illegible] [illegible] (1996).

In conclusion, biodiversity research is based on a blend of natural and [illegible] approaches. In the past [illegible] has been made [illegible] fields need to be [illegible] particular [illegible] approaches [illegible] development of [illegible] models [illegible] [illegible] for estimating and ranking alternative [illegible] and management policies. The following general formulation of an overall system Health Index (HI) has been proposed by Costanza (1992):

$$HI = V \times O \times R$$

where V designates system vigor and represents a measure of the ecological system's activity, metabolism, or primary productivity; O designates system organization and represents a value to one index of the relative degree of organization of the ecological system, including its diversity and connectivity; and R designates system resilience and represents a value to one index of the relative degree of the ecological system's resilience. In other words, an ecological system's overall health is given by its activity weight indices for the system's vigor, organization and resilience. These components are illustrated in Table 2.4.

Table 2.4: Indices of vigor, organization and resilience

Component of health	Related concepts	Related measures	Field of origin	Measurement method
Vigor	Function Productivity Throughput	GPP, NPP, GEP GNP Metabolism	Ecology Economics Biology	Measurement
Organization	Structure Biodiversity	Diversity index Average mutual information predictability	Ecology Biology	Network analysis
Resilience	[illegible]	[illegible]	Ecology [illegible]	Simulation modeling

Source: Costanza (1992).

In addition to the vigor, organization and resilience components, the health index requires the application of different methods of measurement, involving the use of expertise from both the natural and social sciences. One example of an integrated health index, Costanza's [illegible] index, allows for the integrated [illegible] [illegible] of ecosystem health (Costanza, 1992). This index [illegible] the [illegible] of the [illegible]. However, it requires data on all variables occurring in the ecosystem under

Chapter

4

MEASURING BIODIVERSITY

4.1 Introduction

The biological organizational base of our earth is a complex system that comprises an enormous diversity of forms. These are influenced by historical and current situations of both man-made and physical-geographical nature. Traditionally, the measurement of biological diversity is undertaken through the use of genetic, species, and ecosystem richness or variety indices. These concepts will be further envisaged in the present chapter.

4.2 Biotic richness: genetic, species and ecosystem diversity

4.2.1 Genetic diversity

The analysis, conceptualization and measurement of genetic diversity can be done in terms of (1) allelic frequencies; (2) phenotypic traits; and (3) DNA sequences. The same gene can exist in different frequencies or variants. These variants are called alleles. Thus, allelic diversity measures variation in the gene composition of individuals. In general, the more alleles, and the more diverse their frequencies, the greater the genetic diversity. Average expected heterozygosity, the probability that two alleles sampled at random are genetically different, is commonly used as an overall measure. A number of different indices can be applied to measure allelic frequencies' distance (Antonovic 1990).

Phenotypic trait analysis constitutes another approach to measuring genetic diversity. As the name suggests, it proceeds by checking whether individuals share the same phenotype traits. This method avoids the examination of the underlying allelic structure. This method is focused on the measurement of the variance of certain traits and, in general, involves measurable morphological and physiological characteristics of an individual. Since the assessment of allelic frequencies and phenotypic traits variety is often difficult, and comparisons hard and costly to perform, another approach is used. Scientists now use DNA sequence variation to measure genetic variety. The DNA sequence information is obtained through the use of a polymerize chain reaction. For this reason, only a small amount of material, ultimately only a single cell, is required to obtain the DNA sequence data. Closely related species may share up to 95 per cent of their DNA sequences, thus having little diversity in their overall genetic information.

4.2.2 Species diversity

The measurement of species diversity, in its ideal form, consists of a complete catalogue of the distribution and abundance of all species in the area under consideration. However, such an approach is usually not possible unless the area under consideration is small. Therefore, in practice, the measurement of species diversity is often based on samples. The central measures of species diversity are the α, γ and β species diversity, as originally proposed by Whittaker (1960 and 1972). α-diversity refers to the number of species by using only their presence (and not abundance) in a given area. It therefore measures the species richness of a given sample plot (Huston 1994). γ-diversity is often used to assess the overall presence and abundance of species within a large region or at the landscape level (Noss 1983; Franklin 1993). National red species lists can be treated as lower bounds on gamma diversity. β-diversity measures the turnover of species between local areas, i.e., the rate of change in species composition among discrete sites or habitat units (Cody 1986 and 1993). As such, it cannot be expressed in terms of a number of species: it is represented in terms of a ratio, which is interpreted as the species turnover rate. β-diversity is generally used to estimate average changes in species in response to human impact(s) on natural habitats.

Species richness measurement is useful, but it is easily subject to bias. Generally, there is much uncertainty about the total number of species. As a matter of fact, only for very few areas in the world is a complete set of estimates for all species available. Published sources have described circa 1.75 million living species of all kinds of organisms, whereas estimates of the total number of species range around 14 million (Parker 1982; Wilson 1988b; Arnett 2000; CBD 2001). Among the described species, approximately 5,000 are mammals (SI 2002), 10,000 are birds (BLO 2002), 8,000 are reptiles (EMBL 2002), 5,500 are amphibian (AN 2002), and 27,000 are fish (FB 2002). The remaining species consist mostly of invertebrates, including insects, mollusks and crustaceans, plants, fungi and algae. Each species, in turn, contains an immense amount of genetic information. The number of genes ranges from 1,000 in bacteria, through 10,000 in some fungi, up to 30,000 to 40,000 in humans. Moreover, species diversity is associated with habitat scale in a complex way. Thus, one needs to be cautious when comparing species diversity in areas that differ greatly in size.

In addition, species diversity is the result of complex genealogical relationships due to evolutionary history. Alternative species diversity measures supplement species richness with measures of the degree of genealogical difference. Such diversity measures include the weighting of close-to-root species, higher-taxon richness, spanning tree length and taxonomic dispersion (Williams *et al.* 1993). For the time being, practical difficulties with the implementation of such measures have caused a shift in attention towards simpler indicators of species richness. In this context, many ecologists and environmentalists emphasize measurement of species diversity at the community level. Indeed, ecologists often talk with admiration of tropical rain forests and coral reefs as important centers of species richness. The latter can be quantified by using the β-diversity measure as discussed above. Colombia and Kenya, for example, are home to over 1,000 species of birds, while the UK and the east of North America are home to only about 200 species. A coral reef in northern Australia may be home to 500 species, while the rocky shoreline of Japan may be home to only 100 species (UNEP 1995).

4.1.3 Ecosystem diversity

A considerable number of ecologists and environmentalists support the measurement of biodiversity at the organizational level of ecosystem diversity. This is because measuring of biodiversity at this level encompasses a complexity of relationships, both at the intra- and supra-species levels, which play a crucial role in defining the overall distribution and abundance of species. For this reason, there are a number of factors that make the assessment of ecosystem diversity somewhat blurred. Actually, at the ecosystem level many different units of diversity are involved, ranging from the patterns of habitats to the age structure of populations. It is not clear, however, where to draw spatial boundaries. When the ecosystem under consideration is a wetland, for example, it is necessary to look at both aquatic and land components of the landscape. Such an approach may not be equivalent to the aggregate value of system components. In other words, the system may be more than the sum of its parts. This is reflected in the notion of primary value (Gren *et al.* 1994; Turner *et al.* 1997).

The measurement of diversity at the ecosystem function level includes the appraisal of eco-regions or eco-zones, based on the distribution of species, particular physical attributes such as soils and climates, and distinct types of ecosystems (UNEP 1995). The full range of biodiversity values depends on the processes that support the functioning of such large-scale ecological systems. Given the lack of unambiguous biosphere boundaries, different measurement approaches can emerge. One often-used approach is characterized by the definition of bio-geographical provinces. These include, for example, defining the mammals and plant diversity of the arctic tundra and the tropical rain forest, and identifying respective distinct types of ecosystem functions.

4.3 Ecological methods for biodiversity analysis

4.3.1 Overview

This section examines methods and indicators frequently used by ecologists in ecological and diversity assessment. These play a crucial role in guiding policy, since they allow the prediction of an ecosystem's response to alternative management practices and, in this way, can generate a policy ranking. Such rankings are required for selecting and designating nature reserves. For example, in the European Union, 8,819 land sites were designated as nature reserves within the Natura 2000 network, 6,977 within the Habitat Directive (92/43/CEE), and 1,842 within the Birds Directive (79/409/CEE).

A considerable number of approaches are in some way related to Usher's ecological approach to environmental protection (Usher 1989). According to this approach, decision-making is characterized by three steps. First, ecological and diversity attributes are identified and are used to reflect the conservation interests of the respective species or site. Second, criteria are developed for the expression of the attributes in a form that allows evaluation. Finally, values are attached to particular levels of criteria. Red Data Books are an important component of such an assessment.

4.3.2 Red List of threatened species

The Red List was conceived of in order to identify threats to or causes of decline in different species around the world (Fitter and Fitter 1987; Mace and Lande 1991; Mace *et al.* 1992; IUCN 1993; Mace and Stuart 1994). The underlying motivation was (1) to provide information for conservation programs; (2) to assist in the drafting of legislation; and (3) to convey information in a form that is comprehensible to non-specialists. The final goal is to place species into categories according to the threat of their extinction. Eight different categories are used: extinct, extinct in the wild, critically endangered, endangered, vulnerable, lower risk, data deficient and not evaluated (see Figure 4.1). The definition of threatened species is given in Table 4.1. For more details and the criteria referred to below, see the Annex. Since its introduction, the Red List has frequently been used by numerous governmental and non-governmental organizations to guide policy and to help establish conservation priorities. It provides an easily and widely understood method for highlighting which species are under higher risk of extinction. In short, a species is allocated to the Red

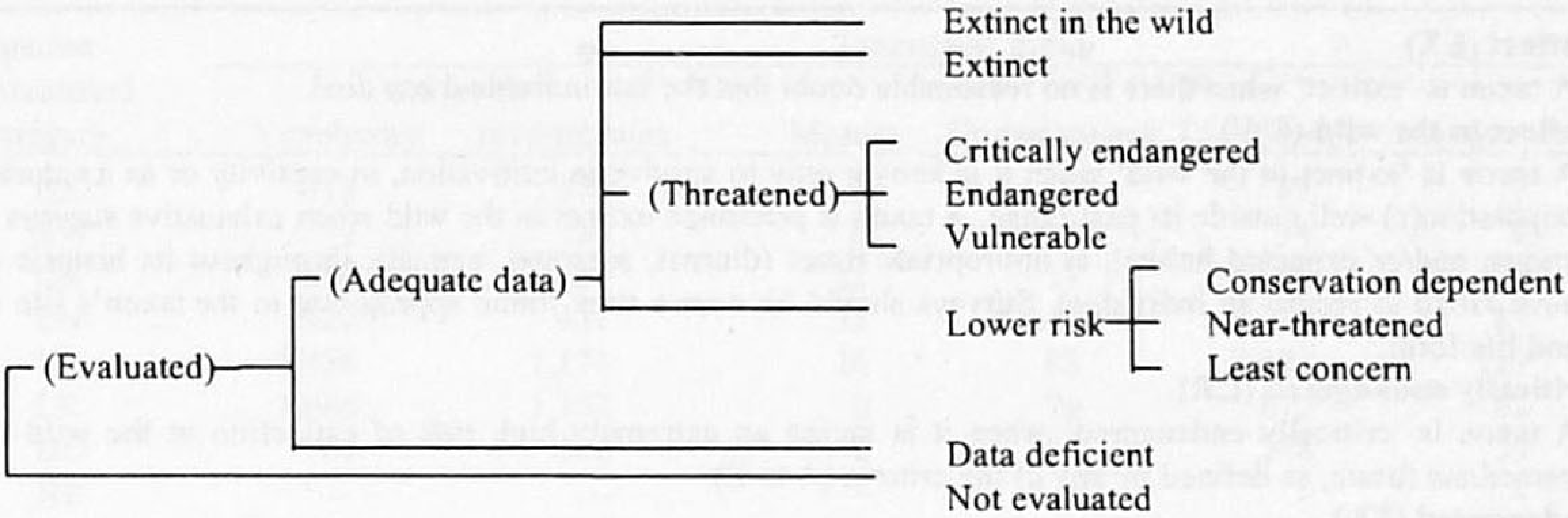

Figure 4.1: Structure of species categories

Source: IUCN (2002).

List only if it meets any one of the criteria as listed in the Appendix (see page 125). At the moment, the IUCN Red List shows more than 16,000 globally threatened species, corresponding to 24 per cent of mammal species and 12 per cent of bird species (see Table 4.2).

However, several problems render category assessment quite difficult in practice. While the classification of a species within a category relies on an objective evaluation, the actual figures used in the definitions of these categories, as described in the Appendix, rely on a subjective process. In practice, the large number of criteria available for defining threatened species in some way reflects the difficulty of conceptualizing these categories. In addition, the Red List of threatened species alone may prove insufficient for guiding priorities for conservation action. In fact, if Red Lists were selected as the ecological method to be used for the establishment of public biodiversity policies, it would be likely that many natural habitats characterized by fertile biotic environments and threatened abiotic systems would be evaluated as not having much value, since this evaluation technique simply provides an assessment of the likelihood of species extinction. Furthermore, the spatial and time discussion of threatened species are not convincingly incorporated in the Red List, so that this information base has to be used with caution in ecosystem policy research. As a result, the application of such a measurement instrument could lead to a (biased) decision not in favor of the protection of the area. Given the scientific understanding of population and ecosystems, it is possible to develop alternative indicators. The following section will focus on these.

Table 4.1: Definition of threatened species

Extinct (EX)

A taxon is 'extinct' when there is no reasonable doubt that the last individual has died.

Extinct in the wild (EW)

A taxon is 'extinct in the wild' when it is known only to survive in cultivation, in captivity or as a naturalized population(s) well outside its past range. A taxon is presumed extinct in the wild when exhaustive surveys in its known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), throughout its historic range have failed to record an individual. Surveys should be over a time frame appropriate to the taxon's life cycle and life form.

Critically endangered (CR)

A taxon is 'critically endangered' when it is facing an extremely high risk of extinction in the wild in the immediate future, as defined by any of the criteria (A to E).

Endangered (EN)

A taxon is 'endangered' when it is not 'critically endangered' but is facing a very high risk of extinction in the wild in the near future, as defined by any of the criteria (A to E).

Vulnerable (VU)

A taxon is 'vulnerable' when it is not 'critically endangered' nor 'endangered' but is facing a high risk of extinction in the wild in the medium-term future, as defined by any of the criteria (A to D).

Lower risk (LR)

A taxon is 'lower risk' when it has been evaluated and does not satisfy the criteria for any of the categories 'critically endangered', 'endangered' or 'vulnerable'. Taxa included in the 'lower risk' category can be separated into three subcategories: 'conservation dependent', 'near threatened', and 'least concern'. The 'conservation dependent' category refers to taxa that are the focus of a continuing specific conservation program targeted towards the taxon in question, where cessation of the program would result in the taxon qualifying for one of the threatened categories above within a period of five years. 'Near threatened' refers to the taxa which do not qualify for 'conservation dependent', but which are close to qualifying for 'vulnerable'. Finally, the 'least concern' taxa category does not qualify for 'conservation dependent' nor 'near threatened'.

Data deficient (DD)

A taxon is 'data deficient' when there is inadequate information to make a direct or indirect assessment of its risk of extinction based on its distribution and/or population status. A taxon in this category may be well studied, and its biology well known, but appropriate data on abundance and/or distribution is lacking. The data listing of taxa in this category indicates that more information is required and acknowledges the possibility that future research will show that threatened classification is appropriate.

Not evaluated (NE)

A taxon is 'not evaluated' when it is has not yet been assessed against the criteria.

Source: IUCN (2002).

4.3.3 Definition of sites of special scientific interest

The first attempt to identify areas of land worthy of legal protection was made by Charles Rothschild in 1915 in the UK. He compiled a list of nature reserves based on his own perceptions as well as those of other leading figures. Since Rothschild did this, methods for selecting sites of special scientific interest (SSSIs) have been much debated. Today, the Nature Conservancy Council (NCC) is the institution in the UK that has the task of identifying 'sites of special scientific interest', defined as any land of special interest by virtue of any of its flora, fauna or geological or physiographic features. The Nature Conservancy Council, according to the system originally developed by Ratcliffe (1977), has identified ten criteria that can be used for identifying conservation priorities, in general, and identifying SSSIs, in particular. These criteria are size, richness (of habitats or species), diversity,

Table 4.2: IUCN Red List of threatened species

Species threatened category	Taxonomic group					
	Mammals			Plants		
	Vertebrates	Invertebrates	Mosses	Gymnosperms	Dicotyledon	Monocotyledon
EX	318	375	3	0	69	1
EW	19	14	0	1	14	2
CR	599	326	22	17	896	79
EN	922	431	32	41	1,110	83
VU	1,986	1,171	26	83	3,093	129
LR	1,666	1,352	0	76	813	62
DD	702	608	0	33	298	39
NE	–	–	–	–	–	–
Total	6,208	3,277	83	251	6,293	395

Source: IUCN (2002).

naturalness, rarity, typicality, fragility, threat, potential value and intrinsic appeal. They are discussed in Table 4.3.

There is no clear guide to the correct criteria to use in any given situation. However, some criteria are more frequently used than others (see Table 4.4). More recently, experts have made use of computer-based systems, with the development of models for population dynamics where one can simulate natural or management changes and estimate the resulting conservation evaluations. These will be discussed in detail in the next section.

4.3.4 Population dynamics of species and ecosystems

Computer modeling techniques have frequently been used to estimate the population dynamics of rare and declining species, which, in turn, are interpreted as a tool for the ecological valuation of alternative habitat conservation scenarios. One specific output of this evaluation approach is the calculation of a minimum viable population, defined as the smallest population that has an acceptable probability of persisting over a given time period (Soulé 1987). This, in turn, allows the calculation of the minimum dynamic area, i.e., the geographic area of suitable habitat required to support the minimum viable population. The minimum dynamic area is a criteria frequently used to aid in conservation evaluation decisions. Recently published studies on the Leadbeater's possum (Lindenmayer *et al.* 1993), the eastern barred bandicoot (Lacy and Clark 1990), the wild boar (Howells and Edward-Jones 1997), and the giant panda (Zhou and Pan 1997) demonstrate the use of population viability analysis.

Several general models have been developed for this task, e.g., VORTEX (Lacy 1993) and METAPOP (Akcakaya 1994), and these can be used to consider the complex interactions between the demographic, environmental and genetic influences on a population. One of the applications of computer systems for ecological evaluation is the System for Evaluating

Table 4.3: Definition of sites of special scientific interest

Size

This criterion refers to the geographical area that the site corresponds to. Generally, the larger the geographical area of the site, the higher its ranking in the conservation evaluation since larger sites generally contain more species than smaller sites, ceteris paribus.

Richness

This measure corresponds to the number of different habitats that exist in a given area. It is generally true that the higher the species richness, the higher the ranking of a site in terms of the conservation evaluation.

Diversity

Diversity measures the number of different types of habitats that exist in a given area as well as the distribution of individuals between the species. According to this measure, the more equitable the distribution of individuals between the species, the more diverse the site.

Naturalness

This criterion conveys an assessment of how disturbed or undisturbed a habitat is. The more natural (less disturbed) the higher its value. The use of this measure implies that there is some natural condition for all habitats. Such a status quo condition is conceptually difficult to measure since humans have in some way influenced almost every habitat in the biosphere.

Rarity

This measure informs us how frequently certain habitats are encountered in a given site.

Typicalness

This measure describes how typical the habitat's assemblages in a given site are for that type of site. Usually, a site is chosen for conservation because it constitutes the best example or representation of a particular habitat.

Fragility

Fragility is the measure that reflects the degree of sensitivity of habitats and species to environmental change. It can be interpreted as the inverse of resilience. Therefore, the more fragile a habitat (a species), the greater the requirement for its protection.

Threat

Threat is a criterion that measures the probability that a habitat will be damaged or destroyed within a given time-scale. Generally, the greater the threat, the more immediate the need to preserve. Rates of loss are usually a good indicator of threat.

Potential value

This is a measure of how, under a given management regime or through natural change, the site will develop features of particular value for conservation. Sites with a good documented history enhance our understanding of the ecological processes involved and, as such, are potentially valuable to science. For example, if the moths in a certain woodland habitat have been recorded for dozen of years, then the sequence of data will provide a good baseline against which the impacts of the management or natural change may be assessed, such as habitat fragmentation in relation to increasing air pollution.

Intrinsic appeal

This last criterion is not a scientific task; but is a normative concept reflecting the moral-socio-economic value structure of the society. For example, humans may tend to give more weight to certain taxonomic groups or habitats. For example, species such as mammals (cuddly), birds (tuneful), wild flowers (colorful) have much public appeal, while beetles, spiders, snakes have less appeal.

Source: NCC (1989).

Rivers for Conservation (SERCON) (Boon *et al.* 1997). The development of SERCON was undertaken by Scottish Natural Heritage, the nature conservation agency in Scotland. The overall objective was to predict the impact of different development scenarios on river ecological conservation value as well as to provide a simple way of communicating the results to planners, developers and policymakers.

Table 4.4: Frequency of use of different criteria in conservation evaluation

Criteria	Frequency of use
Richness (of habitats and species)	16
Naturalness, rarity (of habitats and species)	13
Size	11
Threat of human interference	8
Scenic value	7
Potential scientific value	6
Recorded history	4

Source: Usher (1986).

A final remark is in order here. The past decade has witnessed an enormous increase in the development and use of geographic information systems (GIS), through which the spatial dynamics of multidimensional phenomena can be properly mapped out. Although GIS applications do exist in ecosystem and biodiversity research, there is still no doubt a great potential in the future application of GIS in this field.

Chapter

5

ECONOMIC ANALYSIS OF BIODIVERSITY VALUES

5.1 The concept of economic value

The economic monetary value of a commodity focuses on the welfare of humans. In an economic sense, value is thus the result of an interaction between a human subject and an object like biodiversity. 'Economic value' does not denote an absolute value of levels, but of system changes, preferably marginal or small ones. The reason for this is that the theoretical basis of economic valuation is monetary (income) variation as the response to a certain policy or environmental change. Therefore, the terms 'economic value' and 'welfare change' can, in principle, be used interchangeably.

A basic postulate is that individuals make welfare-optimizing consumption decisions. These decisions are captured in the consumer demand functions with respect to available goods and services. Biodiversity quality considerations enter into these demands. To illustrate this setting, we consider an individual whose utility function has the following form:

$$V = V(x, q, z)$$

Here x is the consumption of the private good, q the quantity of the environmental resource, and z a biodiversity quality indicator. For example, q could represent the number of recreational sites and z the level of species richness. We assume that x is a composite private good whose price is normalized to one. p is the price associated with q. This framework allows the study of a welfare change in the biodiversity quality indicator, z. This change may be interpreted as the introduction of a set of new regulations designed to allow commercial development in the protected areas.

In the original situation, i.e., before the implementation of the new regulation, the individual faces a particular biodiversity quality level. Let us denote such a level by z^0. For a biodiversity quality level z^0, and given the consumer monetary income M, the consumer maximizes $V(x,q,z)$. This yields an optimal consumption bundle (x^0,q^0), with $q^0(p,M,z^0)$ and $x^0(p,M,z^0)$, and a utility level $V^0=V(x^0,q^0,z^0)$. Inserting the demand functions into the utility function gives the indirect utility function $V(x^0(p,M,z^0),q^0(p,M,z^0),z^0)=v(p,M,z^0)$. Table 5.1 summarizes the notation.

Table 5.1: Summary of the results

Variables and function of interest	Original situation	New situation
Biodiversity quality level	z^0	z^1
Utility level	V^0	V^1 with $V^0>V^1$
Indirect utility function	$v(p,M,z^0)$	$v(p,M,z^1)$

The literature suggests two alternative measures that can be used to assess the magnitude of the welfare change as described by the introduction of the new regulation. These are the Hicksian compensating measure and the Hicksian equivalent measure, which are theoretical refinements of the ordinary consumer surplus (Hicks 1943). The Hicksian compensating welfare measure equals the compensating payment, i.e., an offsetting change in income to make the individual indifferent to distinctions between the original situation (status quo) and the new situation. The Hicksian compensating variation *(HC)* is the solution to

$$v(p,M,z^0)=v(p,M+HC,z^1)=V^0$$

i.e., the *HC* measures what must be paid to the individual to make that person indifferent to the new environmental quality level. In other words, under the new situation, the individual's income would be increased by the amount of *HC*, but the person would still be as well off as in the original situation.

The Hicksian equivalent welfare measure corresponds to change in income that would lead to the same utility change as the new situation. The Hicksian equivalent *(HE)* is the solution to

$$v(p,M,z^1)=v(p,M-HE,z^0)=V^1$$

i.e., the *HE* measures the income change that is equivalent to the welfare lost due to the new situation. In other words, that income change is the 'price' that reflects the consumer's maximum willingness to pay (WTP) to avoid an undesirable change in *z*. This interpretation assumes that the benchmark is the level of welfare after the change. If, however, the changes are being compared with the initial situation, then we measure the willingness to accept (WAC). In other words, the two alternative Hicksian welfare measures can be interpreted in terms of the implicit rights and obligations associated with alternative environmental quality levels. In this context, the *HC* carries with it implicitly the assumption that the individual has the use, property and freedom related to the original environmental quality. In contrast, the *HE* is consistent with the idea that the individual has an obligation to accept a reduction in environmental quality. The choice between them is, therefore, ultimately an ethical one. It reflects a value judgment about which underlying distribution of property rights is more equitable (Krutilla 1967). Table 5.2 summarizes the results and the preferred welfare measure, according to the suggestions of the NOAA panel (NOAA 1993).

Table 5.2: Hicksian welfare measures and property rights distribution

Attribute quality	Hicksian equivalent measure (Implied property rights in the change)	Hicksian compensating measure (Implied property rights in the original situation)
Increase	WAC to forgo	WTP to obtain
Decrease	WTP to avoid	WAC to accept

5.2 General aspects of economic valuation of biodiversity

The general context of the economic valuation of biodiversity can be clarified by looking at some of the perspectives included in the discussion in Section 2.3. Economic valuation of biodiversity is based on an instrumental perspective on the value of biodiversity. This means that the value of biodiversity is regarded as the result of an interaction between humans and the object of valuation, which is 'changes in biodiversity'.

Economic valuation provides a monetary indicator of biodiversity values. The reason for this is that the theoretical basis of economic valuation is monetary (income) variation as a compensation or equivalent for direct and indirect impact(s) of a certain biodiversity change on the welfare of humans.

Both direct and indirect values, relating to production, consumption and nonuse values of biodiversity are considered when pursuing an economic valuation of biodiversity. Explicit biodiversity changes, preferably in terms of accurate physical-biological indicators, should be related to these. Biodiversity changes must be marginal or small for economic valuation to make sense.

The economic valuation of biodiversity changes is based on a reductionist approach value. This means that the total economic value is regarded as the result of aggregating various use and nonuse values, reflecting a variety of human motivations, as well as aggregating local values to attain a global value, i.e., a bottom-up approach (see Nunes 2002a; Nunes and Schokkaert 2003).

Moreover, the economic valuation of biodiversity starts from the premise that social values should be based on individual values, independently of whether the individuals are experts in biodiversity-related issues or not. This can be considered consistent with the democratic support of policies. A more detailed discussion and evaluation of monetary biodiversity valuation has to wait until Chapter 6, after the economic valuation applications have been reviewed.

5.3 Motivation for economic valuation of biodiversity

5.3.1 Multiple reasons

The economic valuation of natural resources in general and biodiversity in particular is among the most pressing and challenging issues confronting environmental economics. Major organizations across the world such as the World Bank (1993), Resources for the Future (e.g., Mitchell and Carson 1984; 1993; and Smith 1993; 1994) and OECD (e.g., Pearce and Markandya 1989; Biller and Bark 2001) and US governmental agencies like the Environmental Protection Agency (e.g., McClelland *et al.* 1992) carry out economic value assessment applications. One may wonder for what reason such monetary assessments of environmental resources are undertaken. Four main reasons can be identified. These are performing cost-benefit analysis, environmental accounting, assessing natural resource damage, and carrying out proper pricing. These will subsequently be considered in more detail.

5.3.2 Cost-benefit analysis

Cost-benefit analysis (CBA) is a welfare-theoretic method to trade-off the advantageous and disadvantageous effects of a proposed project by measuring them in monetary terms. CBA emerged as an attempt to systematically incorporate economic information that can be applied to project and policy evaluations. Since CBA has traditionally been defined in terms of gains and losses to society, project-oriented CBA has tended to be confined to public sector investment projects. The first evaluation studies were carried out in the US in the 1950s to deal with 'intangibles' in a consistent way, e.g., for river basin projects and infrastructure projects. These methods found much application, inter alia in World Bank practices. They were also heavily criticized for many inherent shortcomings, which has led to many new or adjusted methods, such as cost-effectiveness analysis, goals-achievement methods and multicriteria analysis (see Nijkamp *et al.* 1991).

The use of CBA to evaluate policy is more recent (see for an overview Hanemann 1992). Like an investment project, policies have costs and benefits. For example, standards for ambient concentrations and taxation of ambient pollutants are two different policies, which, in turn, are associated with different gains and losses to society. The basic rule of CBA in decision-making is to approve any potentially worthwhile policy if the benefits of the policy exceed the costs.

Moreover, to make the best choice, a decision-maker should opt for the policy option with the greatest positive net present value. Other criteria exist, such as ranking and evaluating projects according to their 'internal rate of value' or according to the 'benefit cost ratio' – see Hanley and Spash (1993) for a literature review on CBA and its application to environmental issues. CBA has been used in the US for evaluating policies since the late 1970s. However, only after Reagan's Executive Order 12291, in 1981, has CBA been extensively used for evaluating new regulations. In contrast, in Europe there are no legal requirements for CBA for new regulations. An exception is the UK, whose 1995 Environment Act envisions the use of CBA in policymaking. Clearly, the use of and the critical judgments of CBA in public policy is still a matter of ongoing scientific debate among economists.

5.3.3 Environmental accounting

Various efforts have been made to adjust national accounting systems and associated gross national product (GNP) statistics for the depreciation of environmental assets and for negative externalities such as pollution and the loss of biodiversity. The theoretical literature explores alternative ways of adjusting conventional estimates of national income to reflect environmental deterioration (Aronsson *et al.* 1997). Green accounting is one possible strategy.

The underlying idea is to add to the traditional national accounting system information on physical flows and stocks of environmental goods and services – the so-called physical satellite accounts. In the Dutch context, for example, the Netherlands Central Bureau for Statistics developed the NAMEA, a National Accounting Matrix that includes both economic and Environmental Accounts (Keuning and de Haan 1996). An important aim of green accounting is to obtain an adjusted 'green' GNP. This can play a potentially crucial role in policymaking since the GNP has a powerful influence on macro-economic policy, financial markets and international institutions (OECD, IMF, and World Bank). If national income is wrongly estimated, then economic analysis and policy formulation are based on the wrong premises, thus 'steering' the society by the wrong compass (Hueting 1980; El Serafy 1999). Adjustment of the national accounts to reflect biodiversity loss will lower the GNP (Gerlagh *et al.* 2002).

Nevertheless, practice shows that the adjustment of national accounting systems is not an easy task. It is therefore necessary to achieve international agreement about harmonizing GNP adjustments, allowing for the comparison of GNP and national accounts between countries. Independent of which valuation methods are used for this purpose, it is clear that monetary valuation of the depreciation of environmental assets and negative externalities, such as pollution and the loss of biodiversity, is a key element in green environmental accounting.

5.3.4 Natural resource damage assessment and legal claims

Natural resource damage assessments (NRDAs) appraise how much society values the destruction of natural resources. An important benchmark in the history of NRDA is the massive oil spill due to the grounding of the oil tanker Exxon Valdez in Prince William Sound in the northern part of the Gulf of Alaska on March 24, 1989. This was the largest oil spill from a tanker in US history. More than 1,300 km of coastline were affected and almost 23,000 birds were killed (Carson *et al.* 1992). After the oil spill, the State of Alaska commissioned a legal action in order to assess Exxon's financial liability in the damage to the natural resources. A national Contingent Valuation (CV) study estimated the loss to US citizens as a result of the oil spill. The natural resource damage resulting from the Exxon Valdez oil spill was estimated at \$2.8 billion. For the first time, a governmental decision expressed the legitimacy of nonuse values as a component of the total damage value. To date, NRDAs are only undertaken in the US and have not yet become an issue in Europe because of different legal arrangements.

5.3.5 Proper pricing

Given that most human activities are priced in one way or other, in some decision contexts, the temptation exists to downplay or ignore biodiversity conservation benefits on the basis of non-existence of prices for biodiversity. The simple and simplistic idea here is that a lack of prices is identical to a lack of values. Clearly, this is a slightly biased perspective. The theory of externalities teaches us that many values cannot be incorporated in conventional market transactions. The question is then how to translate such values into monetary dimensions. This is a challenging question to be addressed by economists. An example is the so-called shadow project approach, through which the socially monetary value of an ecosystem that is threatened with extinction is assessed by computing the costs of a similar ecosystem elsewhere. Goeschl and Swanson (2002) discuss alternative biodiversity pricing frameworks and their impacts on the valuation outcome, demonstrating the presence of significant under-estimations of private-based valuation of genetic resources for use in firm R&D when compared to the social value of genetic resources.

5.4 A classification of economic values of biodiversity

It is possible to identify and characterize different value categories of biodiversity. Figure 5.1 shows a classification of biodiversity values that is the basis for the analysis of valuation studies.

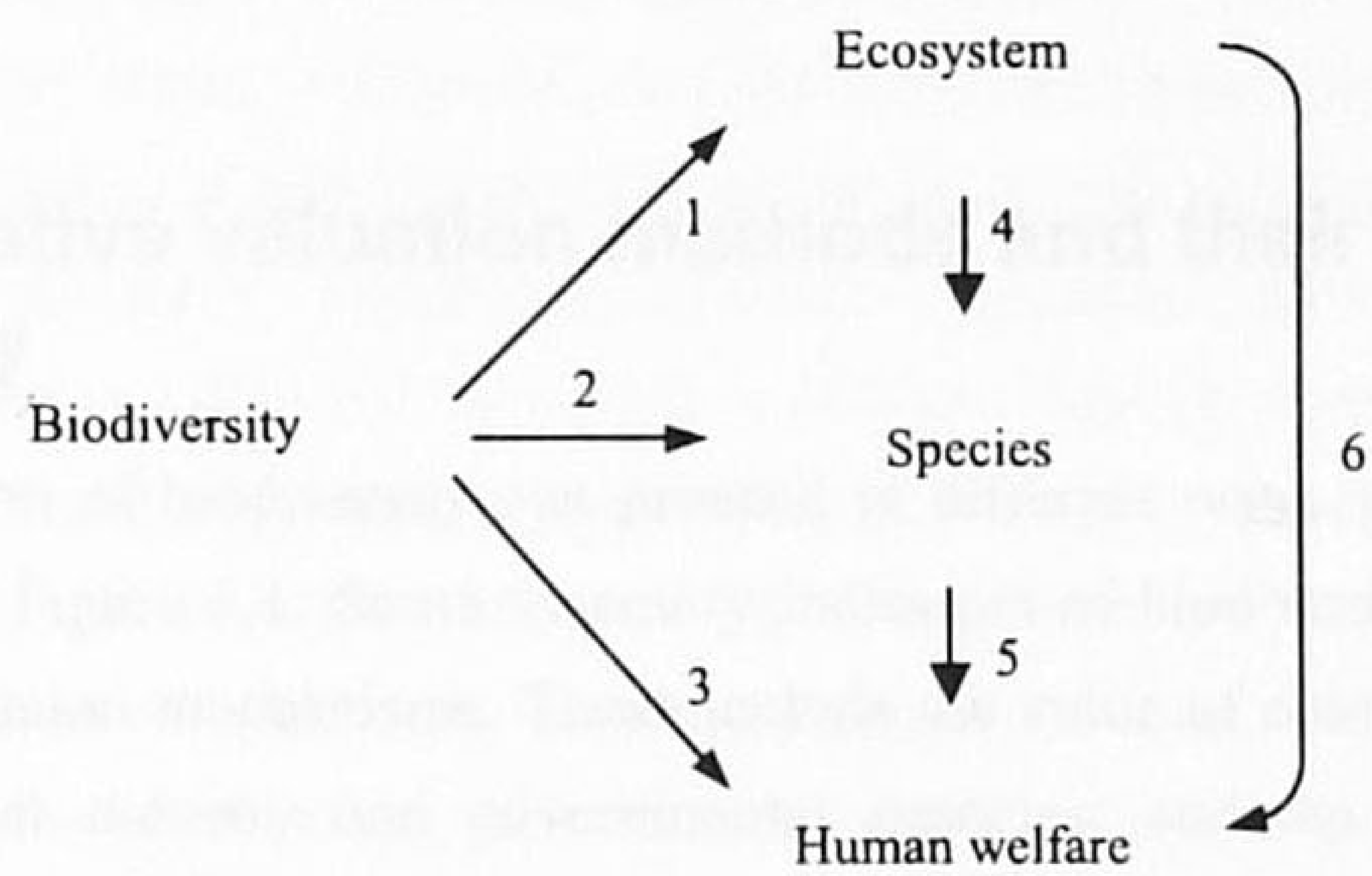

Figure 5.1: Economic values of biodiversity
Source: Nunes and van den Bergh (2001).

A first category, denoted by link 1 → 6, depicts biodiversity benefits that run through ecosystem life support functions and preservation of the ecological structure in natural systems. The diversity of functions generated by ecosystems, in turn, links to the demand for goods and services. This value category can represent, for example, the benefits of flood control, groundwater recharge, nutrient removal, toxic retention, and biodiversity maintenance (Turner *et al.* 2000). A second biodiversity category, denoted by link 1 → 4 → 5, captures the value of biodiversity in terms of natural habitat protection. This can relate, for example, to tourism and outdoor recreational demand.

A third value category, denoted by link 2 → 5, captures the benefits of an overall provision of species diversity. This value category represents the indirect value of biodiversity in biological resources in terms of inputs to the production of market goods. Well-known examples are the pharmaceutical and agriculture industries, which use plant and animal material to develop new medicines and new products (Myers 1988; Simpson *et al.* 1996).

Finally, a fourth category, captured by link 3, denotes a passive or nonuse component of biodiversity value, which reflects moral considerations to other species (bioethics), human philanthropic, or bequest considerations. The latter relates to the knowledge that biodiversity will be available to the next generations.

This chapter has tried to clarify various basic economic aspects of biodiversity evaluation. A solid micro-economic welfare-theoretic foundation seems to be a sine qua non condition for a proper biodiversity evaluation. Nevertheless, in the application field still many advances need to be made. This will be further discussed in Chapter 6.

Chapter 6

MEASURING ECONOMIC VALUE OF BIODIVERSITY BENEFITS

6.1 Alternative valuation methods and their degree of applicability

Monetary valuation of biodiversity can proceed in different ways, partly depending on the paths depicted in Figure 5.1. Some monetary indicators of biodiversity values are based on market price valuation mechanisms. These include the value of contracts recently signed by the pharmaceutical industry and governmental agencies, and the value of the financial revenues related to tourism activities focused on visits to natural areas of high outdoor recreational demand. In the absence of market prices for biodiversity values, which is commonly the case, certain techniques are needed to retrieve consumers' preferences. On the basis of the process through which this happens, one can distinguish two groups of valuation methods: revealed preference methods and stated preference methods. The first type explores the use of existing market data, based on notions of travel cost (TC), hedonic price (HP), averting behavior (AB), and production function (PF) (Mäler 1988; Braden *et al.* 1991). Their common underlying feature is a relationship between a market good and an environmental commodity. For example, when using the travel cost method, researchers estimate the economic value of recreational sites by looking at the generalized travel costs of visiting these sites (Bockstael *et al.* 1991). In contrast, practitioners of the hedonic price method estimate the economic value of an environmental commodity, say, cleaner air, by studying the relation between housing prices and air quality (Palmquist 1991). The averting behavior or production cost function methods is characterized by exploring the relationship of the environmental commodity through a generalized cost function (Cropper and Freeman 1991). For instance, cleaner air can be assessed on the basis of expenditures made to avert or mitigate the adverse effects of air pollution. Avoided cost damage, defensive expenditures, repair costs (or restoration), compensation costs, replacement costs, and relocation costs are specific instances of this method. Finally, the production factor method estimates the economic value of an environmental commodity through the input-output relationship of such a commodity in a production function. In this context, the economic value of cleaner soil is related to the value of increased agricultural output through a dose response method.

The second group of stated preference valuation methods is based on collecting data by means of questionnaires. The best known is contingent valuation methodology (Mitchell and Carson 1989). Indeed, the contingent valuation (CV) method is currently one of the most often used techniques for the valuation of environmental goods. This is partly due to CV features that provide important advantages over revealed preference methods. First, the CV method directly provides the theoretically desirable Hicksian welfare measures. Second, the CV method is the only valuation technique that is capable of shedding light on the monetary valuation of nonuse values, which typically leave no 'behavioral market trace'. Ignoring such values will lead to a systematic bias in the estimation – essentially an underestimation – of the total benefits of biodiversity. Third, CV allows environmental changes to be valued even if they have not yet occurred – ex ante valuation. Therefore, CV offers greater potential scope and flexibility than revealed preference methods. It allows the specification of hypothetical policy scenarios or states of nature that lie outside current or past institutional arrangements or levels of provision. Last but not least, contingent valuation allows one to enrich the information base by submitting the process of value formation to public discussion, which Sen, the 1998 Nobel Laureate in Economics, recognizes as '... an important part of democracy...' (1995, p. 18). As a result, CV can act as an effective tool for policy decision-making.

It ought to be recognized that CV also has some weaknesses. It is based on the hypothesis of full information on the issue concerned, so that respondents are able to offer rational and consistent answers. In reality, such a condition is not always fulfilled. In addition, CV assumes that valuation answers are directed towards the value of an ecosystem resource and 'clean' from strategic behavior or motivational considerations. In empirical studies such considerations are often present. Therefore, economists need to develop an appropriated analytical framework that takes into account the presence of such mixed valuation considerations and estimate the empirical magnitude of the respective mechanisms on the value of the resource under consideration. In this context, Nunes has recently proposed the application of an economic welfare-theoretic foundation for identification, measurement of warm glow motivations and the development of an econometric framework suitable for the empirical assessment of the warm glow valuation mechanism in CV responses (Nunes 2002c).

Table 6.1 shows that certain valuation methods are more appropriate than others for addressing certain types of biodiversity value. For example, revealed preference methods can only be used for a limited number of biodiversity value categories, since they do not allow for a monetary assessment of nonuse values. In contrast, the contingent valuation method is in principle applicable to a multiplicity of biodiversity value categories. However, one needs to recognize that this method will fail for those biodiversity value categories that the general public is not fully informed about or has no experience with. Note in this respect that a questionnaire should be designed comprehensively enough to convey detailed information on changes in ecosystem life support functions and processes related to biodiversity changes so that the latter are not regarded by respondents as too cumbersome. Such information is crucial for obtaining a practical, reliable, and effective questionnaire.

6.2 Biodiversity valuation studies

6.2.1 Overview

The aim of this section is to provide a critical review, rather than a comprehensive survey, of representative biodiversity valuation studies. The discussion is organized as shown in Table 6.1. First, the valuation studies that are reviewed are certainly not exhaustive; they focus predominantly on the assessment of biodiversity value in terms of genetic and species diversity. Subsequently, the discussion is centered on valuation studies that pursue the assessment of biodiversity benefits in terms of natural habitat or ecosystem diversity. Next, we present some valuation study results on biodiversity values linked to the diversity of functions generated by ecosystems, including in particular ecosystem life support functions, flood control and groundwater recharge. Finally, the nonuse or passive value component of biodiversity will be examined as well. The presentation will be kept concise, and the findings will be evaluated in Section 6.3.

6.2.2 Genetic diversity and bioprospecting

Recent years have shown a sharp increase of interest in bioprospecting, i.e., search among the genetic codes contained in living organisms in order to develop chemical compounds of commercial value in agricultural, industrial, or pharmaceutical applications (Simpson *et al.* 1996). This is dominated by pharmaceutical research since most prescribed drugs are derived or patented from natural sources and associated stock of information (Grifo *et al.* 1996; Swanson 1996). This section considers assessments of willingness to pay by the

Table 6.1: Total economic value of biodiversity

Biodiversity value category (see Figure 5.1)	Economic value interpretation	Biodiversity Benefits	Methods for economic valuation (and their applicability)
2 → 5	Genetic and species diversity	Inputs to production processes (e.g., pharmaceutical and agriculture industries)	CV: + TC: – HP: + AB: + PF: + Market contracts: +
1 → 4 → 5	Natural areas and landscape diversity	Provision of natural habitat (e.g., protection of wilderness areas and recreational areas)	CV: + TC: + HP: – AB: – PF: + Tourism revenues: +
1 → 6	Ecosystem functions and ecological services flows	Ecological values (e.g., flood control, nutrient removal, toxic retention and biodiversity maintenance)	CV: – TC: – HP: + AB: + PF: +
3	Nonuse of biodiversity	Existence or moral value (e.g., guarantee that a particular species is kept free from extinction)	CV: + TC: – HP: – AB: – PF: –

Notes: the sign + (–) means that the method is more (less) appropriate to be selected for the design of the valuation context of the biodiversity value category under consideration. See Section 6.1 for an explanation of the abbreviations.

pharmaceutical industries for genetic diversity as input into commercial products. The marginal value of such input, often translated in terms of genetic information for medicinal purposes, is measured by its contribution to the improvement of health care. For example, research by the US National Cancer Institute on screening of plants over the last two decades has yielded various highly effective anti-cancer drugs (e.g., *paclitazel* and *camptothecin*) and anti-leukemia drugs (e.g., *homoharringtonone*) (Cragg *et al.* 1998).

Recent registrations and applications of bioprospecting contracts and agreements between states and pharmaceutical industries represent important benchmarks of monetary indicators for these types of biodiversity values. Estimates are shown in Table 6.2. The most notable of these agreements is the pioneering venture between Merck and Co., the world's largest pharmaceutical firm, and Instituto National de Biodiversidad (INBio) in Costa Rica. At the moment of the contract's signature, in 1991, Merck paid Costa Rica about $1 million and agreed to pay royalties whenever a new commercial product was explored. Since then, INBio has signed contracts on the supply of genetic resources with Bristol-Myers Squibb and other companies and non-profit organizations (ten Kate and Laird 1999; INBio 2001). Another illustration of the market value of genetic diversity is the commercial agreement signed in

Table 6.2: Valuation of bioprospecting agreements

Contractors	Study	Value
INBio & Merck (1991)	2,000 samples of the Costa Rica genetic pool	$1 million
Yellowstone National Park & Diversa (1998)	Thermostable enzyme *Taq* polymerase and bacterium *Thermus aquaticus*	$175,000
Brazilian Extracta & Glaxo Wellcome (1999)	30,000 samples of Brazil biota	$3.2 million

1997 between Diversa, a San Diego-based biotechnology firm, and the US National Park Service. Diversa paid $175,000 for the right to conduct research on heat-resistant microorganisms found in hot springs in Yellowstone National Park (Sonner 1998; Macilwain 1998). More recently, a Brazilian company, Extracta, signed a $3.2 million agreement with Glaxo Wellcome, the world's second-largest pharmaceutical company, to screen 30,000 samples of compounds of plant, fungal and bacterial origin from several regions in the country (Bonalume and Dickson 1999).

Despite the fact that these agreements show a positive economic value of genetic diversity, concern remains about the fairness of such deals. Indeed, some environmental groups have been very critical, claiming that these are unequivocally 'biopiracy' actions (see RAFI 2001). Furthermore, the benefits of the use of genetic diversity in industrial research and technological development may give rise to a number of equity and moral problems (see Swanson 2002). These, however, are not the basis for the pharmaceutical industry's willingness to pay, and therefore are not captured by the market prices of the agreements.

6.2.3 Biodiversity and species preservation

Most of the valuation studies of species preservation have focused on single animal species. Table 6.3 lists some recent studies, all applications in the US, except for a Swedish CV study of wolves (Boman and Bosdedt 1995). The estimates are derived from CV applications and obtained from individual willingness to pay (WTP) to avoid the loss of a particular species. Most welfare gains accrued to individuals are based on recreational activities such as watching threatened or endangered species in their natural habitat, or simply reflect the well-being derived from the knowledge that such a species exists. The later case can be interpreted as relating to nonuse or passive use values. For example, van Kooten (1993) assessed the economic value of waterfowls in a wetland region in Canada; Loomis and Larson (1994) valued 'emblematic' endangered species, namely the gray whale; and Stevens *et al.* (1997) valued the restoration of Atlantic salmon in one river in the state of Massachusetts – see van Kooten and Bulte (2000) for more examples.

Table 6.3: Valuation of single species

Author(s)	Study	Mean WTP estimates (per household/year)
Stevens *et al.* (1997)	Restoration of the Atlantic salmon in one river, Massachusetts	$14.38 to $21.40
Jakobsson and Dragun (1996a)	Conservation of the Leadbeater's possum, Australia	$29 (Australian $)
Boman and Bostedt (1995)	Conservation of the wolf in Sweden	700 SEK to 900 SEK
Loomis and Larson (1994)	Conservation of the gray whale, US	$16 to $18
Loomis and Helfand (1993)	Conservation of various single species, US	From $13 for the sea turtle to $25 for the bald eagle
van Kooten (1993)	Conservation of waterfowl habitat in wetlands region, Canada	$50 to $60 (per acre)
Bowker and Stoll (1988)	Conservation of the Whooping Crane	$21 to $141
Boyle and Bishop (1987)	Conservation of the bald eagle and the striped shiner, Wisconsin	From $5 for the striped shiner to $28 for the bald eagle
Brookshire, Eubanks and Randall (1983)	Conservation of the grizzly bear and the bighorn sheep, Wyoming	From $10 for the grizzly bear to $16 for the bighorn sheep

Alternatively, economists can pursue valuation studies of species preservation that focus on more than one species, as shown in Table 6.4. The estimates are higher than the single species value estimates, though not as high as one would expect, bearing in mind the initial single species estimates. For example, the WTP of the wolf study in Sweden alone corresponds to more than 70 per cent of the WTP for 300 Swedish endangered species. An interpretation of such estimation results may be, however, heavily criticized because of the CV's design and execution (see Carson 1997). Nevertheless, some authors prefer to work with other categories of biodiversity value, namely value categories related to natural habitat, ecosystem functions and services flows protection. These are discussed in the following sections.

6.2.4 Biodiversity and natural habitat preservation

A problem with the interpretation of the value estimates of species preservation is the frequently missing link between the value assigned to a particular (set of) species and the area needed to protect (their) habitats. Some studies instead link the value of biodiversity to the value of natural habitat conservation. Some examples are listed in Table 6.5. For example, Bateman *et al.* (1992) undertook a contingent valuation study to assess the monetary value of conserving the Norfolk Broads, a wetland site in the UK that covers three National Nature Reserves. The estimation results from a mail survey show that respondents living in a zone defined as 'near-Norfolk Broads' had a WTP of £12, whereas those living in the 'elsewhere

Table 6.4: Valuation of multiple species

Author(s)	Study	Mean WTP estimates (per household/year)
Jakobsson and Dragun (1996b)	Preservation of all endangered species in Victoria	$118 (Australian $)
Desvousges *et al.* (1993)	Conservation of the migratory waterfowl in the Central Flyway	$59 to $71
Whitehead (1993)	Conservation program for coastal nongame wildlife	$15
Duffield and Patterson (1992)	Conservation of fisheries in Montana Rivers	$2 to $4 (for residents) $12 to $17 (for non residents)
Halstead *et al.* (1992)	Preservation of the bald eagle, coyote and wild turkey in New England	$15
Hampicke *et al.* (1991)	Preservation of endangered species in West Germany	140 DM to 250 DM
Johansson (1989)	Preservation of 300 endangered species in Sweden	1,275 SEK
Samples and Hollyer (1989)	Preservation of the monk seal and humpback whale	$9.6 to $13.8
Hageman (1985)	Preservation of threatened and endangered species populations in the US	$17.73 to $23.95

UK' zone had a WTP of £4. In the context of the Netherlands, Hoevenagel (1994) asked 127 respondents for an annual contribution to a fund from which farmers in the Dutch meadow region would receive a government grant if they managed their land in a way that enhances wildlife habitat. The average WTP was between NLG 16 and NLG 45. Brouwer (1995) found similar results.

More recently, Nunes (2002b; 2002c) used for the first time a national CV application in Portugal to assess willingness to pay for the protection of natural parks and wilderness areas. The mean WTP results ranged from $40 to $51. In a US context, Mitchell and Carson (1984) used the CV method to value the preservation of water ecosystems and the aquatic-related benefits provided by all the rivers and lakes in the US. Loomis (1989) used CV to value the preservation of Mono Lake, California – see valuation figures in Table 6.5. Kealy and Turner (1993) estimated the benefits derived from the preservation of the Adirondack aquatic system. The WTP estimates ranged between $12 and $18. Boyle (1990) valued the preservation of the Illinois Beach Nature Reserve. The estimation results show that the average WTP ranged between $37 and $41. Silberman *et al.* (1992) studied the existence value of beach ecosystems for users and nonusers of New Jersey beaches. The results show that the mean WTP for a user is about $15.1 while the mean WTP for a nonuser is about $9.26.

Other studies link the value of biodiversity to the value of protection of natural areas with high tourism and outdoor recreational demand. In this biodiversity value category, biodiversity has been assessed by various methods, including contingent valuation, the travel

Table 6.5: Valuation of natural habitats

Author(s)	Study	Mean WTP estimates (per household)
Nunes (2002b, c)	Protection of natural parks and wilderness areas, Portugal	$40 to $51
Wiestra (1996)	Protection of ecological agricultural fields, The Netherlands	NLG 35 (single-bounded)
Richer (1995)	Desert protection in California, US	$101
Brouwer (1995)	Protection of peat meadow land, The Netherlands	NLG 28 to NLG 72
Carson *et al.* (1994)	Protection of the Kakadu conservation zone and National Park, Australia	$52 (minor impact scenario) $80 (major impact scenario)
Hoevenagel (1994)	Enhancing wildlife habitat in the Dutch peat meadow region, The Netherlands	NLG 16 to NLG 46
Kealy and Turner (1993)	Preservation of the aquatic system in the Adirondack region, US	$12 to $18
Hoehn and Loomis (1993)	Enhancing wetlands and habitat in San Joaquin Valley in California, US	$96 to $184 (single program)
Diamond *et al.* (1993)	Protection of wilderness areas in Colorado, Idaho, Montana and Wyoming, US	$29 to $66
Silberman *et al.* (1992)	Protection of beach ecosystems, New Jersey, US	$9.26 to $15.1
Bateman *et al.* (1992)	Protection of a wetland site, the Norfolk Broads, UK	£4 to £12
Boyle (1990)	Preservation of the Illinois Beach State Nature Reserve, US	$37 to $41
Loomis (1989)	Preservation of the Mono Lake, California, US	$4 to $11
Smith and Desvousges (1986)	Preservation of water quality in the Monongahela River Basin, US	$21 to $58 (for users) $14 to $53 (for nonusers)
Bennett (1984)	Protection of the Nadgee Nature Reserve, Australia	$27
Mitchell and Carson (1984)	Preservation of water quality for all rivers and lakes, US	$242
Walsh *et al.* (1984)	Protection of wilderness areas in Colorado, US	$32

cost method and market prices such as tourism revenues. Some examples are as listed in Table 6.6.

For example, the World Tourism Organization (WTO 1997) estimated that Ecuador earned $255 million from ecotourism in 1995. A major sum accrued to a single park, the Galapagos Islands. In Rwanda, gorilla tourism in the Volcanoes National Park generated directed revenues of $1.02 million annually until 1994, or $68 per ha (AG Ökotourismus/BMZ 1995).

Studies of less popular parks indicate lower values. The recreational value of Mantadia National Park in Madagascar was estimated to range between $9 and $25 per ha (Mercer *et al.* 1995). One particularly interesting valuation result is shown in the study by Norton and Southey (1995). This study calculates the economic value of natural habitat for biodiversity protection in Kenya by assessing the associated opportunity costs of foregone agricultural

Table 6.6: Valuation of tourism and outdoor recreation

Author(s)	Study	Measurement method	Estimates
Moons (1999)	Enjoyment received in forest-related recreational activities in Flanders, Belgium	Travel cost	BEF 1,030 per trip
Chase *et al.* (1998)	Protection of recreation opportunities in three National Parks, Costa Rica	Contingent valuation	\$21.60 to \$24.90 per visitor
WTO (1997)	Ecotourism in Ecuador	Tourism revenue	\$255 million annually
Layman *et al.* (1996)	Chinook salmon in the Gulkana river, Alaska	Travel cost	\$17 to \$60 per trip
AG Ökotourismus (1995)	Gorilla tourism in Volcanoes National Park, Rwanda	Tourism revenue	\$1.02 million annually
Mercer *et al.* (1995)	Recreational value of Mantadia National Park, Madagascar	Tourism revenue	\$9 and \$25 per ha
Norton and Southey (1995)	Biodiversity conservation in Kenya	Production function	\$203 million annually

production, which is estimated to be \$203 million. This is much higher than the \$42 million of net financial revenue from wildlife tourism. Layman *et al.* (1996) explored the travel cost method to estimate the recreational fishing value of Chinook salmon in the Gulkana river, Alaska. The estimates of the mean consumer surplus per day range from \$17 to \$60 for actual trips, depending upon the wage rate.

More recently, Chase *et al.* (1998) studied ecotourism demand in Costa Rica. The value estimates result from a survey of foreign visitors to three national parks: Volcan Irazu, Volcan Poas, and Manuel Antonio. Manuel Antonio National Park registered the highest WTP, \$24.90. Finally, Moons (1999) used the travel cost method to assess the economic value of recreational activities in the Meerdal-Heverlee forest in Belgium.

6.2.5 Biodiversity and ecosystem functions and services flows

The CV method has been widely used for valuing biodiversity benefits around the world, in terms of both species diversity and natural habitat protection. Nevertheless, when it comes to the monetary valuation of ecosystem functions, CV may not always be the best choice. This is because ecosystem functions, such as ecosystem life support, are not an issue that the general public is familiar with. In addition, the complexity of the relationships involved makes their accurate and comprehensive description in a survey extremely difficult. Researchers frequently end up using valuation methods based on travel costs, averting behavior or production functions. In this context, valuation studies based on soil and wind erosion, water quality, and wetland ecosystem functions can be distinguished. These are listed in Table 6.7.

Table 6.7: Valuation of ecosystem functions and services

Author(s)	Study	Measurement Method	Estimates
Choe *et al.* (1996)	Value of a public health program at Times Beach, Philippines	Travel cost	$1.44 to $2.04 per trip
Laughland *et al.* (1996)	Value of a water supply in Milesburg, Pennsylvania	Averting expenditures	$14 and $36 per household
Turner *et al.* (1995)	Life support value of a wetland ecosystem on a Swedish island, Baltic Sea	Replacement costs	$0.4 to $1.2 million
Barbier (1994)	Preservation of Hadejia-Jama'are wetlands, Nigeria	Production function	N 850 to N 1,280 per ha
Pina (1994)	Spending of ecotourists in Mexico	Travel cost	$60 to $100 per day
Abdalla *et al.* (1992)	Groundwater ecosystem in Perkasie, Pennsylvania	Averting expenditures	$61,313 to $131,334
McClelland *et al.* (1992)	Protection of groundwater program, US	Contingent valuation	$7 to $22
Andreasson-Gren (1991)	Nitrogen purification capacity of a Swedish island in the Baltic, Gotland	Replacement costs	SEK 968 per kg
Torell *et al.* (1990)	Water in-storage in the High Plains aquifer	Production function	$9.5 to $1.09 per acre-foot
Tobias and Mendelsohn (1990)	Tourism and ecotourism based on non-consumptive uses of wildlife in Costa Rica	Tourism revenue	$1.2 million per ha
Ribaudo (1989a, b)	Water quality benefits in ten regions in the US	Averting expenditure	$4.4 billion
Huszar (1989)	Value of wind erosion costs to households in New Mexico	Replacement costs	$454 million per year
King and Sinden (1988)	Value of soil conservation in the farm land market of Manilla Shire, Australia	Hedonic price	$2.28 per ha
Holmes (1988)	Value of the impact of water turbidity due to soil erosion on the water treatment	Replacement costs	$35 to $661 million annually
Walker and Young (1986)	Value of soil erosion on (loss of) agricultural revenue in the Palouse region	Production method	$4 to 6$ per acre
Veloz *et al.* (1985)	Soil erosion control program in a watershed in the Dominican Republic	Production function	DR$ 260 per ha

6.2.5.1 Soil and wind erosion valuation studies

One category of valuation of ecosystem functions and services relates to soil erosion. Veloz *et al.* (1985) performed an economic analysis and valuation of soil conservation in the Dominican Republic. They estimate that, for a 25-year land use interval, the net returns with the introduction of erosion control programs are about DR$ 260 per hectare. Walker and Young (1986) estimate the damage caused by soil erosion in terms of (loss of) agricultural revenue in the Palouse region of northern Idaho and western Washington to be equal to $4 and $6 per acre, for scenarios with slow and rapid technological progress, respectively. Holmes (1988) studied the impact of water turbidity due to soil erosion on the costs incurred by the water treatment industry. Estimates show that mitigation costs ranged from $4 to $82 per million gallons of water for conventional and direct filtration systems, respectively. When applying these estimates to the American Water Works Association figures on total surface water withdrawal, the nationwide damages induced by turbidity are estimated to be between \$35 and \$661 million annually. King and Sinden (1988) have explored the use of the hedonic price method in order to capture the value of soil conservation in the farmland market of Manilla Shire, Australia. The hedonic land market price regression results show that soil condition (e.g., depth of topsoil) has an implicit marginal price of \$2.28/ha. More recently, Huszar (1989) studied erosion due to wind in New Mexico. According to this study, wind erosion costs to households are due to increased cleaning, maintenance and replacement expenditures, and also to reduced consumption and production opportunities. A household cost function was estimated on the basis of 242 survey respondents. The total household costs were estimated to be \$454 million per year.

6.2.5.2 Water quality valuation studies

Water quality has been valued in many studies. Ribaudo (1989a, b) is responsible for one of the most comprehensive studies of valuing water quality benefits. The author valued the economic benefits from a reduction in the discharge of pollutants in waterway systems for nine impact categories: recreational fishing, navigation, water storage, irrigation ditches, water treatment, industrial water use, steam cooling, and flooding. The study targeted all of US territory, which was operationalized in terms of ten regions (Appalachia, Corn Belt, Delta, Lake States, Mountain, Northeast, Northern Plains, Southern Plains, Pacific and Southeast). Benefits were defined in terms of changes in defensive expenditures, changes in production costs, and changes in consumer surplus, depending on the damage category and the availability of data. The total water quality benefits were estimated to be \$4.4 billion.

Torell *et al.* (1990) has assessed the market value of the water in storage in the High Plains aquifer, a water ecosystem that underlies parts of Colorado, Kansas, Nebraska, New Mexico, Oklahoma, South Dakota, Texas, and Wyoming. Water value estimates range from \$9.5 per acre-foot in New Mexico to \$1.09 per acre-foot in Oklahoma. Abdalla *et al.* (1992) has conducted an economic valuation of the contamination of a groundwater ecosystem in Perkasie, Pennsylvania. The study was conducted with the help of a household survey that asked for information about respondents' expenditures since December 1987, the time when the contamination was first detected. The average weekly increase in averting expenditures per household, among those that undertook averting actions in response to the contamination, was \$0.40. The costs of these actions, when extrapolated for Perkasie's total population, ranged from \$61,313 to \$131,334, depending on the wage rate used to reflect the value of lost leisure time.

More recently, Laughland *et al.* (1996) have assessed the economic value of the water supply in Milesburg, also in Pennsylvania. The authors used cost savings with two alternative values of time and the contingent valuation method. The mean averted cost ranged between \$14 and \$36, using family income and using the minimum wage to value time, respectively. Finally, Choe *et al.* (1996) have estimated the economic benefits of surface water improvements through a public health pollution program at Times Beach, in the Philippines. Welfare estimates ranged from \$1.44 to \$2.04 per trip.

6.2.5.3 Wetland ecosystem function valuation studies

Andreasson-Gren (1991) estimated the benefit of nitrogen abatement due to wetland restoration by estimating the replacement costs for conventional nitrogen abatement technologies. The nitrogen purification capacity of wetlands was estimated for Gotland, a Swedish island in the Baltic Sea. According to the study's results, the total value of a marginal increase in nitrogen abatement in Gotland was about SEK 968 per Kg. Barbier (1994) conducted a value assessment of the Hadejia-Jama'are wetlands, Nigeria, by focusing on the opportunity costs of its loss. The valuation analysis covered direct use values of the floodplain to the local population through crop production, fuelwood and fishing. The present value of the aggregate stream of such benefits was estimated to be in the range of 850 to 1,280 Naire per ha. Turner *et al.* (1995) addressed the problem of valuation of wetland ecosystems. This study also attempted to break down direct and indirect value into a much finer set of categories. Their valuation, based on Folke (1991), refers to the assessment of the life support value of Martebo mire, a wetland ecosystem on a Swedish island in the Baltic Sea. An annual monetary estimate of the replacement cost was derived from information about the amount of industrial energy needed to substitute for the loss of wetland-produced goods and services.

6.3 Discussion of the valuation results

From a review of the economic valuation studies it is clear that the assessment of biodiversity values does not lead to a univocal, unambiguous monetary indicator. Instead, the range of monetary estimates of biodiversity values is expected to depend on the level of life diversity under consideration, the biodiversity value type under assessment, and the valuation method being employed. A summary of the various combinations of possible elements is presented in Table 6.8.

At the most basic level of life diversity, the market value of bioprospecting contracts signed between the pharmaceutical and agricultural industries and governmental agencies sheds some light on the economic value of genetic diversity. However, these industries' willingness to pay only considers the potential impact of genetic diversity from the use of plant and animal material in the development of new medicines and new products.

Table 6.8: Synthesis of the valuation results

Life diversity level	Biodiversity value type	Value ranges	Method(s) selected
Genetic and species	Bioprospecting	From: $175,000 To: $3.2 million	Market Contracts
	Single species	From: $5 To: $126	Contingent valuation
	Multiple species	From: $18 To: $194	Contingent valuation
Ecosystems and natural habitat	Terrestrial habitat (non use)	From: $27 To: $101	Contingent valuation
	Coastal habitat (non use)	From: $9 To: $51	Contingent valuation
	Wetland habitat (non use)	From: $8 To: $96	Contingent valuation
	Natural areas habitat (recreation)	From: $23/trip To: $23 million/year	Travel cost, tourism revenues
Ecosystems and functional	Wetland life support	From: $0.4 million To: $1.2 million	Replacement costs
	Soil and wind erosion protection	Up to $454 million/year	Replacement costs, hedonic price, production function
	Water quality	Up to $661 million/year	Replacement costs, averting expenditure

Indirect, existence, and moral values of genetic diversity are not included in the contract market value. Therefore, and at best, these contracts should be interpreted as providing the lower bounds of the economic value of genetic diversity changes. A more extreme position is adopted by some environmental groups, which interpret these market agreements as unequivocal biopiracy actions that cannot serve as the basis for genetic diversity values. Alternatively, one can pursue an economic valuation of biodiversity at the species level. The application of economic valuation to species diversity can be operationalized in terms of single and multiple species studies. The respective value range estimates are characterized by a high degree of uncertainty. For example, willingness to pay is higher in multiple species studies than in single species studies, though not so high as one would initially expect. This reflects not only the complexity of accurately assessing species distinctions, and genetic distances, but also the difficulty of dealing with the substitutability of species. If, for example, a single species valuation study fails adequately to consider that other species are possible substitutes then it may have limited relevance for the valuation of species diversity. Nevertheless, recent research efforts focusing on improving the methods of economic valuation, especially after the NOAA panel recommendations for contingent valuation, have contributed to more accurate survey design. Contingent valuation can thus contribute to the assessment of economic values of species diversity.

Chapter

7

INTEGRATED ECOLOGICAL-ECONOMIC MODELING AND ANALYSIS OF BIODIVERSITY

7.1 Introduction

The analysis and modeling of biodiversity are rooted in the domains of the natural and social sciences; they require the study of human economic activities, their relationships with biodiversity, and with the structure and functions of ecosystems. The combination or integration of the two approaches implies in practice often a somewhat qualitative, formal, sequentially integrated framework. Interdisciplinary work involves economists or ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific purposes. This approach's underlying objective is to develop a common way of thinking about the modeling and valuation of biodiversity. This may require activities such as reduction, simplifying or summarizing. This section provides a survey of frameworks and methods of integrated ecological-economic modeling and the valuation of biodiversity. It ends with an illustration of a regional integrating modeling exercise.

7.2 Frameworks and theories underlying integrated modeling

Before discussing specific methods and models it seems useful to say a few words about the frameworks and conceptual perspectives underlying the integration of economics, ecology and other disciplines. The literature shows various examples of such simple frameworks.

Surveys are offered by Barbier (1990), van den Bergh and Nijkamp (1991), van den Bergh (1996), Costanza *et al.* (1997), Ayres *et al.* (1999) and Turner *et al.* (2000).

A very general and almost non-theoretical ('no assumptions') framework is the Driver-Pressure-State-Impact-Response (DPSIR) framework, a variation on the framework proposed for environmental data classification by Turner *et al.* (2000) and Rotmans and de Vries (1997) for integrated analysis and modeling. The components can be interpreted as follows:

- 'Driver' = economic and social activities and processes;
- 'Pressure' = pressures on the human (health) and environmental system (resources and ecosystems);
- 'State' = the physical, chemical and biological changes in the biosphere, human population, resources and artifacts (buildings, infrastructure, machines);
- 'Impact' = the social, economic and ecological impacts of natural or human-induced changes in the biosphere;
- 'Responses' = human interventions on the level of drivers (prevention, changing behavior), pressures (mitigation), states (relocation) or impacts (restoration, health care).

According to Rotmans and de Vries (1997) integration can be of various types. Vertical integration means that the causal chain in the PSIR or DPSIR framework is completely described in a model ('Close the PSIR loop'; p. 25). Horizontal integration (of subsystems) in this context is defined as the coupling of various global biogeochemical cycles and earth system compartments (atmosphere, terrestrial biosphere, hydrosphere, lithosphere and cryosphere). Full or total integration means a combination, leading to the complex linking, of various drivers, pressures, states, impacts and responses, thus allowing for various synergies and feedback. The integration frameworks proposed in environmental and ecological economics represent more specific theoretical choices than the DSPIR model. We discuss several in the following section.

7.3 Integrated model assessment

A very general method of developing integrated models is the systems approach (also 'systems dynamics'). This includes a wide range of model types: linear versus nonlinear, continuous versus discrete, deterministic versus stochastic, and optimizing versus descriptive. The systems approach allows us to deal with concepts like dynamic processes, feedback mechanisms, and control strategies (see Bennett and Chorley 1978; Costanza *et al.* 1993). One can integrate two subsystems, or have a hierarchy or nesting of systems. The fixed elements in the system can either be considered black boxes or be described as empirical or logical processes themselves. The systems approach is suitable for integrating existing models, and can incorporate temporal as well as spatial processes.

Costanza *et al.* (1993) distinguish between economic, ecological and integrated approaches on the basis of whether they optimize: (1) generality, characterized by simple theoretical or conceptual models that aggregate, caricature and exaggerate; (2) precision, characterized by statistical, short-term, partial, static or linear models with one element examined in much detail; and (3) realism, characterized by causal, nonlinear, dynamic-evolutionary, and complex models.

These three criteria are usually conflicting, so a trade-off between them is inevitable. A distinction between analytical and heuristic integration is relevant here. Analytical integration means combining all aspects studied in a single model (and therefore model type). Heuristic integration can proceed by using the output of one model as input to another, and vice versa, as well as by extending this through (finite) iterative interaction. In this case different model types, such as optimization models and descriptive models, can be combined. If one desires to attain a great deal of analytical power, analytical integration seems attractive, whereas striving for realism would imply the use of a heuristically linked set of models from different disciplines. Striving for empirically sound models often implies modest approaches to improving precision, which usually goes at the cost of model use in a wider context or with a wider range of parameter values. The development of integrated models, through the joint efforts of economists and ecologists, is based on bringing together elements, theories or models from each discipline and transforming these for the purpose of integration. This may require operations such as reduction, simplification or summarizing. The results may not always be greeted with enthusiasm within the disciplines, especially when they neglect certain nuances or different viewpoints.

Many integrated models defined at the level of ecosystems are based on the standard systems-ecological approach (Patten 1971; Jørgenson 1992). They include ecosystem modules that describe the effects of environmental pollution, resource use and other types of disturbance. A main problem is modeling the effects of multiple stress factors, since the empirical basis for this is often lacking. Various integrated models have been developed for terrestrial and aquatic systems. Surveys are presented in Braat and van Lierop (1987), van den Bergh (1996) and Costanza *et al.* (1997). Some studies have paid much attention to spatial aspects, focusing on spatial disaggregation into zones (for instance, Giaoutzi and Nijkamp, 1995; van den Bergh and Nijkamp 1994) or on land use planning in interaction with landscape ecology (see Bockstael *et al.* 1995). Formal theoretical approaches in ecology that provide a basis for these approaches have been described by Watt (1968), Maynard-Smith

(1974), Roughgarden *et al.* (1989) and Jørgenson (1992). Perrings and Walker (1997) consider resilience in a simple integrated model of fire occurrences in semi-arid rangelands such as those found in Australia. The model describes the interaction between extreme events (fire, flood, and droughts), grazing pressure, and multiple locally stable states. Carpenter *et al.* (1999) develop and explore water and land use options in an integrated model of a prototypical region with a lake that is being polluted. This model combines rationally bounded behavior, supposedly in accordance with the reality of regional resource and environmental management, and a nonlinear ecosystem module describing processes occurring at different speeds. The model generates multiple locally stable states as well as 'flipping' behavior (see also Janssen *et al.* 1999). Swallow (1994) integrates theoretical models of renewable and non-renewable resources to address multiple uses and tradeoffs in wetland systems. A special category of integrated modeling is sometimes referred to as the biophysical or energy approach. This aims to integrate economic and environmental ecological processes in energy-physical dimensions, based on the notion that any system is constrained by energy availability (Odum, 1983). These models include energy and mass balances. A central concept in this approach is 'embodied energy', which is defined as the direct and indirect energy required to produce organized material structures. Applications of these energy-inspired models cover ecosystems, economic systems, and environment-economy models (Odum, 1987). An extended application to a regional system is presented by Jansson and Zucchetto (1978) (see also Zuchetto and Jansson 1985).

The recent focus on integrated assessment of the enhanced greenhouse effect (potential climate change) can be regarded as the new wave in 'world models', where (again) economists and others have tended to rely on different model approaches (Bruce *et al.* 1995). The integrated climate assessment models integrate results from the natural sciences (physics, chemistry, biology, earth sciences) and the social sciences (economics, sociology, political science), and have so far given rise to a continuation of the trend in world models towards increasing detail and disaggregation. These climate assessment models have a multilayered conceptual structure that distinguishes physical and environmental effects of human activities from adjustments to climate change by humans (individuals, firms, organizations) and policy responses (mitigation, aimed at the causes) at various spatial levels (Parry and Carter 1998).

7.4 Specific methods and models

Integrated models can have different formats. Table 7.1 illustrates some characteristics of integrated models and provides general examples. One important distinction is between policy optimization and evaluation (usually numerical simulation) models. One of the first

Table 7.1: Characterizing integrated models

Model criterion	Range of choice	Examples of distinct approaches
Analytical integration	Optimization (benevolent decision maker); Equilibrium (partial or general); Game-theoretical; Dynamic-mechanistic; Adaptive (multi-agent & dynamic); Evolutionary (irreversible, bounded rationality)	Many theoretical models: growth theory, renewable resource economics (fisheries, forestry, water quality/quantity), systems models (limits-to-growth, Meadows), cost-effectiveness models (RAINS), welfare optimization (DICE)
Heuristic integration	Satellite principle; Multilayer subsystems; Sequential; Parallel consistent scenarios; Aggregation of Indicators; Evaluation	Regional environmental quality models (Resources for the Future), world models (Club of Rome), integrated assessment model, the present study
Spatial coverage	World; National; Regional; Urban, Local, Ecosystem	Ecosystem modeling, macroeconomic modeling, regional modeling, urban modeling, world models
Spatial disaggregation	Single region; Multiregion; Spatial grid (GIS)	Integrated assessment models (climate change), land use models
Aggregation level	Micro (individuals, households); Macro (national economy, main sectors, global); Sectorial; Interest groups; Homogeneous land plots; Spatial grids; Temporal (days, seasons, years)	Computable general equilibrium models, macroeconomic models (Keynesian), multisector models, land use models, landscape models

Source: van den Bergh (2000).

and famous integrated assessment models used in policymaking is the model RAINS (Alcamo *et al.* 1990). This includes an optimization algorithm for calculating cost-effective acidification strategies in Europe, aimed at realizing deposition targets throughout Europe, and taking account of sensitive natural areas (forests and lakes). This model is a rare case of direct science-policy influence, as it was used in the negotiations on transboundary air pollution in Europe. Castells (1999) offers an informative analysis of the institutional and evolutionary dimensions of the interaction between scientists, research institutions and negotiations on international environmental agreements, with special attention given to the RAINS model and the acid rain context in Europe.

In the area of integrated assessment models for CO_2 emissions (climate) strategies, one can find both economic optimization (Nordhaus 1994) and detailed descriptive model systems like IMAGE and TARGETS (Alcamo 1994; and Rotmans and de Vries 1997). DICE by Nordhaus (1991) is the first example of a policy optimization model for climate change. The model essentially combines economic growth theory with a simplified climate change model. Tol (1998) provides a short account of the evolution of the economic optimization approach to climate change research. He emphasizes the attention placed upon the analysis of uncertainty and learning from a cost-effectiveness perspective, which has given rise to various model formulations and analyses. More recently, Janssen (1998) and van Ierland (1999) present informative surveys and categorizations of macroeconomic-cum-environment and macro-level integrated models, including the climate-oriented integrated assessment models. Van Ierland devotes special attention to the various 'regionalized world' models (with acronyms like RICE, CETA, MERGE, DIALOGUE, FUND). Van den Bergh and Hofkes (1998) have collected distinct approaches to integrated modeling with an economic emphasis that focus on sustainable development questions in theory and in practice, as well as at global and regional levels.

7.5 Interaction between integrated modeling and monetary valuation

Progress in improving models to provide economic information, particularly predictive information, will require vital and dynamic interdisciplinary dialogue. At this level, integrating modeling and monetary valuation can present important advantages for guiding policy by presenting important interactions. First, values estimated in a valuation study can be used as parameter values in model studies. Benefits or value transfer (e.g., meta-analysis exercises) can be used to translate value estimates into other contexts, conditions, locations or temporal settings that do not allow for direct valuation in 'primary studies' (due to technical or financial constraints). Second, models can be used to generate values under particular scenarios. In particular, dynamic models can be used to generate a flow of benefits over time and to compute the net present value, which can serve as a value relating to a particular scenario of ecosystem change or management. Third, models can be used to generate detailed scenarios that enter valuation experiments. An input scenario can describe general environmental change, regional development or ecosystem management. This can be fed into a model calculation, which in turn can provide an output scenario with more detailed spatial or temporal information. The latter can then serve, for example, as a hypothetical scenario for valuation, which is presented to respondents in a certain format (graphs, tables, story, diagrams, pictures) so as to inform them about potential consequences of the general policy or exogenous change. Computer software can be used in such a process. Finally, the outputs of model and valuation studies can be compared. For instance, when studying a scenario for wetland transformation, one can model consequences in multiple dimensions (physical, ecological and costs/benefits), and aggregate these via a multi-criteria evaluation procedure, with weights being set by a decision-maker or a representative panel of stakeholders.

Alternatively, one can ask respondents to provide value estimates, such as willingness to pay for not experiencing the change. If such information is available for multiple management scenarios, then rankings based on different approaches can be compared.

7.6 Advantages and disadvantages of integrated modeling

Using integrated economic-environmental models for the analysis and evaluation of biodiversity issues has both advantages and disadvantages. Three main advantages are: (1) handling data, information, theories, and empirical findings from various contributing disciplines in a systematic and consistent way; (2) being explicit about assumptions, theories and facts; and (3) addressing complex phenomena, interactions, feedback, laborious calculations and temporally, spatially and sectorally detailed and disaggregate processes. An argument against non-formal approaches to integrated research is that these fail to provide for a systematic and consistent linking of data, theories and empirical insights from various disciplines. Instead, these approaches tend to result in a battle of perspectives based on distinct and usually implicit premises and information bases. Models force researchers to be explicit about at least the latter two inputs to integrated research. Most of the disadvantages of integrated modeling apply to non-model-based integrated research as well. They include: an unclear synergy of approximations and uncertainties; rough application of monodisciplinary theories and empirical insights; simplification of complex phenomena (e.g., by treating them as a black box); misinterpretation or arbitrary choice of disciplinary perspectives by the model, and a lack of systematic or complete linking of subsystems or submodels. Complex or high-dimensional models have the extra disadvantage of being difficult to calibrate and validate, and of lacking transparency.

The main disadvantage of models perhaps is that they are trusted too much, so that they run the risk of being interpreted as objective representations of reality, and are then taken too seriously, especially by laypersons and policymakers. On the other hand, policymakers often indicate their doubts about formal modeling. Shackley (1997) states that numerical models have, despite their long tradition of development and widespread use, not achieved the epistemological status that the controlled laboratory experiment has in the natural sciences (and more recently in the social sciences and in environmental economics in particular; see Settle *et al.* 2002). This relates to the fact that modeling results never 'prove' anything, since they do not generate real or physical processes. The best way to view theoretical and especially empirical models is to consider them tools for hypothetical experiments with complex systems, which serve as analogies or pictures of real-world systems that do not allow – technically, morally or politically – for experimentation. In other words, complex model systems, notably integrated economic-ecological models, are heuristic devices for learning about the real-world system, rather than for predicting its real course of behavior. In addition, integrated modeling is restricted by the model type.

If economic and ecological models fit within a (general) systems framework, then they may be blended into a single model structure, where compartments or modules may represent the original models, and certain outputs of one module serve as input for another. Nevertheless, it is often not easy to link models directly. For instance, if both the economic and ecological systems are represented in the form of programming or optimization models, several options are available: look for a new, aggregate objective; adopt a multiobjective or conflict analysis framework; or, when possible, derive multiple sets of optimality conditions and solve these simultaneously. Alternatively, when the economic and ecological systems are represented by different model types, it is difficult to suggest how they could be linked to one another. When economic models have an optimization format and ecosystem models have a descriptive format, direct technical integration seems feasible; otherwise heuristic approaches are needed. In section 7.7 we will offer an empirical illustration of an integrated modeling application.

7.7 A spatial-economic-hydroecological model for biodiversity evaluation

7.7.1 Background

Wetlands are very sensitive ecosystems, generally characterized by a high level of biodiversity, and the ability to provide a wide range of goods and services to human societies as well as to generate nonuse values. Many wetlands are subject to much stress from human activities. Reducing the stress on wetlands requires a spatial matching between physical planning, hydrological and ecological processes, and economic activities. Policy design with such an aim can benefit from spatially integrated modeling. In a study for the Vecht area in the Netherlands a triple layer model was developed that integrates information and concepts from the social and natural sciences through a set of linked spatial hydrological, ecological and economic models, formulated at the level of grids and polders. This study illustrates the possible use of plant species diversity information in integrated economic-environmental modeling. To our knowledge, similar approaches seem to be non-existent or extremely rare in the literature (for a complete account of the study, see van den Bergh *et al.* 2001a, b).

The Vecht area is the floodplain of the river Vecht, located in the center of the Netherlands. The main activities incorporated in the system of models are housing, infrastructure, agriculture, recreation and nature conservation. The formulation of alternative development scenarios is dominated by land use and land cover options that are consistent with the stimulation of agriculture, nature or recreation. Two aggregate performance indicators have been constructed from model output, namely net present value of changes and environmental quality, the latter incorporating elements of biodiversity. The spatial characteristics of these indicators are retained in a spatial evaluation that ranks scenarios.

The study approach followed is based on explicit spatial scenario formulation, modeling and evaluation. A valuation study was considered too difficult to perform, given the size and heterogeneity of the area. Valuation studies seem more suitable for smaller, more homogeneous areas (Gren *et al.* 1994; Turner *et al.* 1998).

7.7.2 Description of the area

7.7.2.1 Current situation

The Vecht area is located between the river Vecht to the west, the sandy ice-formed hill ridge 't Gooi circa 8 km to the east, the Randmeer (part of a large lake which formerly was connected to the sea) to the north, and the city of Utrecht to the south (see Figure 7.1). The area is a wetland with many shallow lakes and fens interspersed with agricultural fields. It overlaps two provinces (North Holland and Utrecht), has a size of 262.6 km^2, which includes ten municipalities, and a population of 212,839. Large urban centers lie just outside the area. Agriculture and nature are the main land uses in the area; industry is virtually absent.

Most nature areas are owned by nature conservation organizations, both governmental and non-governmental. The variety in natural and artificial lakes, reedlands, marshes, grasslands and alder forests creates a mixture of different succession phases characteristic of this type of wetland, as well as a mixture of landscapes. The value of nature in this area is high. The whole area is part of the Dutch Ecological Network and one of its lakes, Naardermeer, is listed as a Ramsar wetland. This value is also reflected in intensive outdoor recreation, including sailing, camping, walking and cycling. In particular, many people from the nearby cities of Utrecht, Amsterdam and Hilversum are regular visitors to the area.

7.7.2.2 Threats to the wetlands

The problems currently faced by the Vecht wetlands relate to its hydrology, chemistry and physical planning. The balance between surface water and groundwater has changed. Groundwater and rainwater are the original sources of water for the wetlands. During the 1970s, 20 million m^3 of drinking water were abstracted annually from the hill ridge, reducing the input of groundwater substantially and resulting in lower water tables and less seepage into the wetlands. To compensate for high levels of evapotranspiration in the summer and to minimize soil subsidence as a result of mineralization of peat soils with their exposure to air,

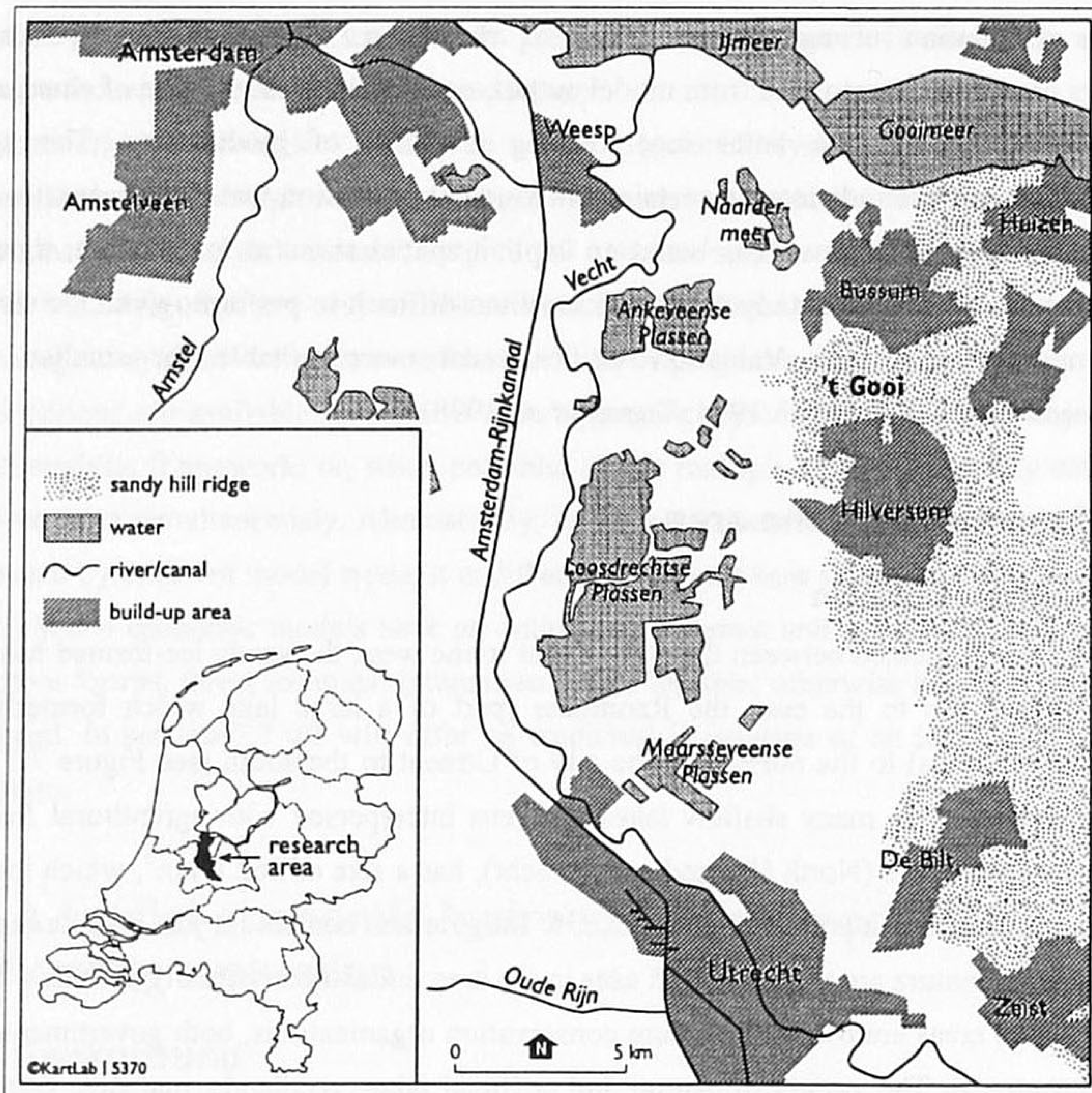

Figure 7.1: The research area and its surroundings

Source: van den Bergh *et al.* (2001b)

surface water from the river Vecht was allowed to enter the area. Further, the two reclaimed lakes (Bethune and Horstermeer), with their artificially low water levels, have effectively reversed the direction of local water flows.

A second problem, partly related to the changes discussed above, is that of water chemistry. The wetlands are suffering from nutrient enrichment as a result of a number of factors: the penetration of nutrient-rich water from the river Vecht into the wetlands, intensified agriculture, local sewerage treatment plants, mineralization of peat soils, and outflow from illegal waste-dumps. Algal blooms and a deterioration of the quality of nature in the area have been recorded. A third problem is pressure from the spatial pattern of human activities. Recreation especially has been intensifying. Attempts by agriculture to intensify have met with mixed success due to technical restrictions on water tables and physical planning regulations to protect nature.

7.7.2.3 Development scenarios

Development scenarios for the Vecht area that could be tested by models and subsequently evaluated were formulated. These scenarios reflect choices made in physical planning, nature policy, agricultural policy, and regulation of recreation, i.e., the main political and economic interests in the area. The scenarios are in line with present policy. In particular, the nature and recreation scenarios attempt to improve environmental quality in a corridor that runs north-south through the study area. The scenarios are spatially disaggregated and are formulated at the level of grids and polders. This contributes to both accuracy and realism of the descriptions. The hydrological parameters are defined at a grid level (500 m x 500 m) and the economic parameters at a polder level (on average 200 ha). This is related to the fact that the hydrological model assumes homogeneity at the level of grids, and the economic model at the level of polders. Any information at a grid level can be easily aggregated to the polder level. Given that the integrated modeling approach is static, the scenarios are also static. They will be used in a comparative static analysis, where changes from a reference (or base) scenario are compared with alternative scenarios. Each scenario specifies land use and water levels at the polder level for the 73 polders that comprise the study area.

The four scenarios for development of the Vecht area are the following: (1) reference (base or business-as-usual); (2) stimulation of agriculture; (3) stimulation of nature; and (4) stimulation of recreation. These scenarios take the present conditions into account. Scenarios 2 to 4 focus on core human activities in the region, and allow comparison of quite distinct, although still realistic, possible future organizations of the Vecht area. Note that the 'agricultural scenario' is very similar to the 'reference scenario' because the present situation is largely the product of actions to realize good agricultural performance.

7.7.3 Method of integrated research

The method of integrated study addressed integration at three levels, as shown in Figure 7.2. The first level involves the formulation of development scenarios for the Vecht area that include consistent settings for the hydrological and economic parameters. The second level entails the integration of hydrological quality and quantity modeling, vegetation response modeling, and economic modeling and accounting. The hydrological-ecological part of the modeling and analysis was performed on a grid basis, and subsequently aggregated to the level of polders, to make it consistent with the economic analysis. Integration at this level includes explicit links between the natural science and economic models. A third integration

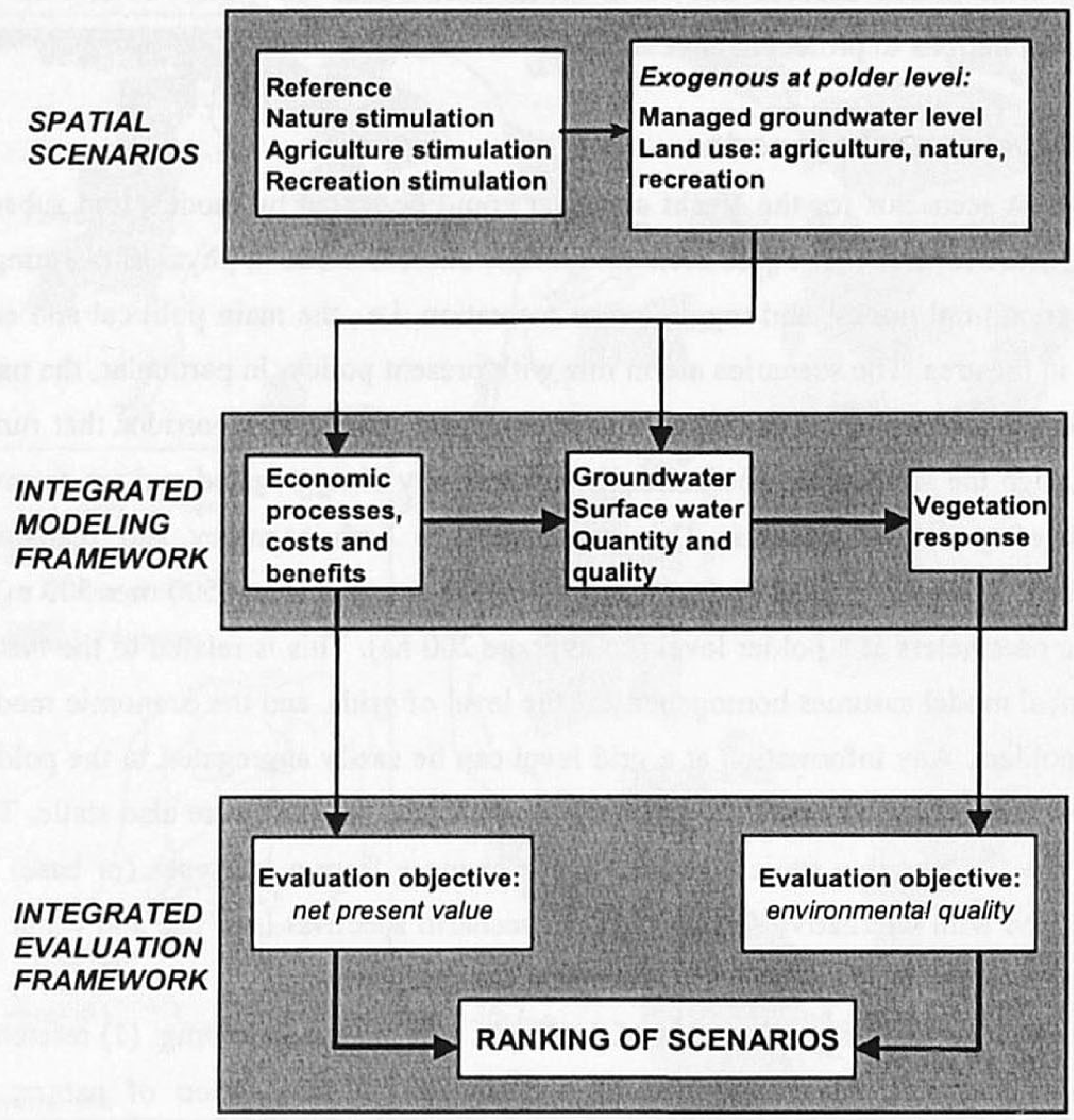

Figure 7.2: The integrated approach adopted in the 'Vecht area' case study

level consists of two steps. The first one aggregates output from the models into performance indicators reflecting objectives in the evaluation. The full study generated three performance indicators (see van den Bergh *et al.* 2001a). Here we present only two: net present value and environmental quality. The final and essential step in the integration is the evaluation procedure in which these performance indicators were used to rank scenarios. Two objectives drive this evaluation: economic efficiency and environmental quality. The latter attempts to capture ecological criteria describing how well wetland ecosystems are functioning.

The ecological model that was used is ICHORS (Barendregt *et al.* 1993). It predicts the occurrence of wetland plants on the basis of statistical relationships between the conditions in the environment (a total of 25 variables) and the presence of 250 plant species. The statistical relations in ICHORS originate from data sets with hundreds of samples collected from all types of wetlands throughout the region. The environmental conditions comprise soil type, land use management, groundwater level, and concentrations of major ions and nutrients in groundwater or in surface water. The model estimates the probability that each wetland plant species will be found at a given site. Since environmental conditions can result from specific management options or external events and trends, the model is suitable to estimate the effects of scenarios.

The economic model was developed on the basis of two aims:

1. Present economic indicators for each scenario and polder, to enter the multi-criteria evaluation procedure, and perform financial cost-benefit analysis; the economic indicators calculated are the changes in net present value and employment, and;
2. Calculate the changes in the run-off of nutrient flows under each scenario, to enter into the eco-hydrological model; this covers environmental indicators for the run-off and surplus of nitrogen and phosphate.

The economic model focuses attention on financial cost benefit analysis and indicators, i.e., it only considers financial transactions and excludes the (social) costs and benefits for which no market price is paid, such as those related to changes in nature. The reason is that this study also presents separate ecological indicators that measure improvements in environmental quality. In the evaluation step economic and ecological indicators are combined. This means that if the environmental improvements were to be incorporated in the economic indicators the evaluation would represent a sort of double-counting.

In order to perform the above task a spatial economic model was formulated that describes agriculture, nature conservation and outdoor recreation. The inputs to the spatial-economic model consist of the settings under specific scenarios as defined above; economic, agricultural and environmental data on a hectare level for the various scenarios and settings; and environmental quality indicators per polder as calculated by the model (see below).

7.7.4 Results

The natural science modeling gave rise to the following insights. Two opposing influences exist on water quality and subsequently on the presence of the desired plant species. First, drainage attracts fresh groundwater from surrounding areas, positively affecting wetland species. Second, maintaining high water tables inevitably requires the inflow of water from the river Vecht, which is rich in nutrients and has a relatively high salinity level; this has a negative effect on wetland species. The net effect of these processes is also influenced by: acidification resulting from the infiltration of rainwater; drainage with subsequent mineralization of peat and release of additional nutrients; and intensification of agriculture with extra input of manure and fertilizers. As a result of these partly conflicting processes, stimulating nature will only be possible in specific parts of the region.

Table 7.2 gives a concise overview of the results obtained with the economic model. It shows the net present value (NPV) of changes aggregated per sub-region (North, Middle, South and Total). The NPV of the agricultural scenario is calculated from the increased benefits of intensifying agriculture in various polders. The intensification of agriculture has a

Table 7.2: The net present value per region under the three scenarios (*)

Region	Agricultural scenario	Nature scenario	Recreation scenario
North	76.99	– 109.02	698.73
Middle	35.00	– 34.20	741.61
South	63.60	– 83.16	542.29
Total	175.59	– 226.38	1,982.65

(*) Changes in million €, relative to the reference scenario.

positive effect on the NPV, because the benefits per hectare of agricultural land increase.

The construction of an index of environmental quality was based on the output of the vegetation response model and, to a lesser extent, of the hydrological model. Output from the former consists of the probability of occurrence of some 250 plant species per 500 m x 500 m grid cell. At a theoretical level, indicators of environmental quality attempt to capture three ecosystem characteristics: process, structure, and resilience. These aggregate characteristics were taken to be represented by: process as reflected by the presence of species typical of nutrient-rich conditions, structure as the presence of species typical of fen and peatlands, and resilience as the presence of species suggesting that the natural succession series in these wetlands has been diverted.

A selection of plant species per category was made on the basis of expert judgment. Four indicators (eutrophication, peat-formation, diversity and non-resilience) were then calculated per grid cell as the average probability for the selected species per scenario. Grid cell values for these indicators were then aggregated to derive polder values. Further, the eutrophication and non-resilience indicators were given a negative value since they make negative contributions to environmental quality. The indicators were then combined to derive an index of environmental quality. In this combination, it was assumed that the three ecosystem characteristics contribute equally to environmental quality, and that the eutrophic and peat-forming indicators contribute equally to ecosystem process.

Finally, there is need to assess the different policy options by means of a comparative research method. In an evaluation aimed at ranking the scenarios a combination of multicriteria and spatial evaluation (see van Herwijnen 1999; van Herwijnen and Rietveld 1999) within the software package DEFINITE (Janssen and van Herwijnen 1994) was used. The DEFINITE software is based on a set of systematically organized methods (see Chapter 8) through which the most appropriated choice probability can be identified. Earlier studies (e.g., Barendregt *et al.* 1992; Bos and van den Bergh 2002) have focused on cost-benefit analysis of specific scenarios and measures for the Vechtstreek. The objectives of the evaluation were: economic efficiency (approximated by NPV) and environmental quality (approximated by the index described above). The DEFINITE software gives a relative evaluation of policy choices in terms of a performance score based on multidimensional assessments. Some findings, based on this particular evaluation approach, are summarized in Table 7.3. From a welfare perspective, the ranking of the scenarios, from most to least preferred, is clearly: Recreation > Nature > Agriculture > Reference. It should be noted that a range of evaluation approaches was examined. From this, it was concluded that maintaining spatial detail in the evaluation for as long as possible can help to address uncertainty and sensitivity of outcomes and parameter settings. Aggregation can average out spatial differences within and among scenarios. Moreover, the pattern of aggregation can influence the ranking of alternatives. Space matters, especially in studies of wetlands, because hydrological processes are inherently spatial (see for more details and results, van den Bergh *et al.* 2001a and b).

Table 7.3: Derivation of a single welfare score per scenario

Performance indicator	Unit	Reference	Agriculture	Nature	Recreation
Net present value	NPV (million €)	0	175.6	– 226.4	1,982.6
Environmental quality	Index [0,100]	0	0.24	7.24	6.22
Welfare	Index [0,100]	5	11	50	93

This section has elucidated the use of evaluation exercises for biodiversity research. Multicriteria methods turned out to be helpful tools in this respect. The next chapter will address these methods in more detail.

Finally, there is a need to assess the different policy options by means of a comparative research method. In an evaluation aimed at ranking the scenarios a combination of multicriteria and spatial evaluation (see van Herwijnen 1999, van Herwijnen and Rietveld 1999) with the software package DEFINITE (Janssen and van Herwijnen 1996) was used. The DEFINITE software is based on a set of evaluation or comparison methods (see Chapter 8) through which the most appropriate choice or alternative can be identified. Earlier studies (e.g., Bamdregt et al. 1992, Bos and van den Bergh 2002) have focused on cost-benefit analysis of specific scenarios and measures for the Vechtstreek. The objectives of the evaluation were: economic efficiency (approximated by NPV) and environmental quality (approximated by the index described above). The DEFINITE software gives a relative evaluation of policy choices in terms of a performance score based on multi-dimensional assessments. Some findings, based on this particular evaluation approach, are summarised in Table 7.3. From a welfare perspective, the ranking of the scenarios, from most to least preferred, is Nature, Recreation > Nature > Agriculture > Reference. It should be noted that a range of evaluation approaches was examined. From this it was concluded that maintaining spatial detail in the evaluation for as long as possible can help to address uncertainty and sensitivity of outcomes and evaluation. Although aggregation into 'average' outcomes misses differences within and among scenarios. Moreover, the pattern of aggregation considerations, the ranking of alternatives, spatial outputs, especially in studies of wetlands, because hydrological processes are intrinsically spatial (see for more detail on this topic, van den Bergh et al. 2001b and 2004).

Table 7.3. Comparison of the policy options

Evaluation indicator	Unit	Optimum	Agriculture	Nature	Recreation
Net present value	MFlr (million f)	0	12.5	1261	1,782.6
Environmental quality	Index (0,100)	9	0.84	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

This section has illustrated the use of combining ecological and multicriteria methods. Multicriteria methods turned out to be helpful tools in this respect. The next chapter will address these methods in more detail.

Chapter 8

MULTICRITERIA EVALUATION

8.1 Introduction

Biodiversity is a compound, multifaceted concept comprising ecological, economic, geographic and social dimensions. The often unpriced and intangible nature of biodiversity makes it difficult to incorporate this concept in a directly measurable and straightforward way into economic assessment methods. From a methodological perspective we face two difficult issues. First, biodiversity cannot be characterized by a single indicator, but requires a multidimensional set of characteristic attributes, each of which has a site-specific and object-specific impact on biodiversity. In the second place, in most evaluation problems biodiversity is not the sole judgment criterion but has to be considered in a broader trade-off context of relevant policy angles, such as socio-economic, industrial, and demographic ones. Consequently, the clear-cut economic assessment of biodiversity is an almost insurmountable task and hence the use of cost-benefit analysis – as a comprehensive evaluation tool – is problematic in most real-world cases.

An important analytical tool that has gained much popularity in the framework of multidimensional evaluation analysis is multicriteria analysis. This has developed into a toolbox for complex assessment problems and has seen a wide range of applications in environmental studies because of its potential to consider simultaneously different mutually irreducible or incompatible judgment criteria (see, for example, Nijkamp and Vindigni 2000). As it also allows for the inclusion of monetary aspects of an evaluation problem, it can also been seen as a generalized and more flexible version of cost-benefit analysis. Multicriteria evaluation can be regarded as a more general method of welfare analysis, and as such presents a good alternative to traditional, neo-classical economic analysis of social welfare and policy instruments (see Munda, 1997).

8.2 Multicriteria decision models

Multicriteria analysis is part of decision theory, which aims to identify the best possible alternative out of a set of rival choice options, where each option is characterized by multiple different judgment criteria. These criteria are mutually conflicting in nature and hence lead to complex trade-off problems. Furthermore, the relative policy priority or weight attached to each individual criterion impacts the final choice. Another feature of multicriteria analysis is that the impacts of each choice and the weights assigned to each criterion may be measured on various scales, ranging from cardinal to binary. The measurement level and the weight generation procedure clearly have a consequence for the type of multicriteria technique to be used (see also Nijkamp *et al.* 1991). We will give a concise formal introduction into multicriteria decision-making, starting from a decision framework with a single criterion.

In decision-making, the comparison of two actions a and b, often involving the consideration of several points of view with conflicting characters, will be encapsulated into one of the following choice possibilities: a is preferred to b $(a \succ b)$; b is preferred to a $(b \succ a)$; a and b are indifferent $(a \approx b)$; and a and b are incomparable $(a ? b)$ – see Nijkamp and Vindigni (2001). A usual decision-making approach is characterized by the maximization of an objective function g, defined in terms of each action. In this context, the decision-maker's preferences meet the following model:

$$\forall\, a,b: \qquad a \succ b \;\Leftrightarrow\; g(a) > g(b)$$

and

$$a \approx b \Leftrightarrow g(a) = g(b)$$

This model is incompatible with the existence of a minimum threshold below which the decision-maker feels indifferent to both of the two actions. In this context, with the introduction of a positive threshold q, the model of comparison is as follows:

$$\forall\, a,b: \qquad a \succ b \;\Leftrightarrow\; g(a) > g(b) + q$$

and

$$a \approx b \Leftrightarrow |g(a) - g(b)| \leq q$$

where $|\;|$ is the distance measure.

Finally, if the decision-maker feels indifferent between the two actions according to a variable threshold, the decision-maker's preferences meet the following model:

$$\forall\, a,b: \qquad a \succ b \Leftrightarrow g(a) > g(b) + q(b)$$
and
$$a \approx b \Leftrightarrow |g(a) - g(b)| \leq q(b)$$

This model presents a decision-maker with a set of unique decision criteria, which are supposed to capture all the relevant choice aspects of the problem. Such a mono-criterion comparison can be interpreted as based on 'global preferences', i.e., preferences based on a choice model that simultaneously takes all relevant points of view into account. In a multicriteria framework these should be interpreted as 'partial preferences', i.e., preferences based on a choice model that takes a specific point of view with regard to the respective criterion. In a multicriteria framework, different conflicting evaluation criteria are taken into consideration. Therefore, there is a priori no action that is unambiguously better than all the others from the points of view of all the criteria considered simultaneously. In other words, in a multicriterion framework there is no unambiguous objective solution.

Multicriteria decision methods constitute an important toolbox for helping a decision-maker to master actions involving multiple criteria (Arrow and Raynaud 1986; Roy 1990). A multicriteria problem is a situation in which the definition of a set of actions and a family of criteria on the set of actions is possible with the aim: (1) to determine a subset of actions considered to be the best with respect to some given criteria – choice problem; (2) to divide the set actions into subsets according to some pre-specified norms – sorting problem; and (3) to rank the set of actions from the best to the worst – ranking problem.

An evaluation method can support the ranking of alternative decisions regarding management, policy, development scenarios or projects. This can lead to a complete ranking, the best alternative, a set of acceptable alternatives, or an incomplete ranking. In general, multiple criteria are relevant in the evaluation procedure. Each of these should be comprehensive and measurable. Comprehensive means that the criterion's value is sufficiently indicative of the degree to which a specific objective is met. Measurable means that it is consistent with a particular measurement scale that allows ordering of the alternatives (management strategies, policies, scenarios, and projects) for a particular objective. The ratio, interval ordinal and binary scales qualify as suitable scales of measurement, as they allow the ordering of alternatives. Over the past decades, a wide range of multicriteria analysis (MCA) methods has been developed. A classification of the most frequently used techniques can be based on the alternatives selected, on one hand, and on the type of information used, on the other hand. According to the former, the methods can be distinguished into continuous methods and discrete methods, corresponding to an infinite and finite number of alternatives, respectively. According to the latter, the methods can be divided into three classes: numerical quantitative methods, if the measurement of the information is on an interval or ratio scale; qualitative methods, if information is measured on a nominal scale or ordinal scale; and mixed methods, if the measurement uses both types of information. Table 8.1 shows the various steps that can be followed in defining the structure of a multicriteria decision problem.

In empirical research we have observed a wide range of different methods that can be applied for evaluation purposes. A concise overview can be found in Nijkamp *et al.* (1991). Their suitability depends, among other things, on the level of information or data precision they can handle and the way priorities (weights) are included. Nowadays there are standard software packages available that can readily be used for the application of MCA - see the DEFINITE method indicated in Chapter 7. Various methods have been developed and applied to the context of environmental and ecological economics issues (Janssen 1992; Munda 1995; van den Bergh 1996; and Janssen and Munda 1999). Examples of applications of MCA in the field of biodiversity can be found in Giaoutzi and Nijkamp (1995). This study deploys a computer simulation model to identify feasible sustainable development pathways for the Greek Sporades Islands. MCA methods are then used to select the most desirable development scenario based on trading off the conflicts between economic progress and ecological sustainability. Other illustrations of MCA are discussed in the next sub-section.

8.3 Practical applications to biodiversity issues

In practical applications two types of input are required for a multicriteria analysis: a table with effects scores for each alternative and criterion (or indicator) and a set of weights. Sensitivity analysis can focus on either of those. Many ecological-economic analyses are based on dynamic and spatial models that generate temporal and spatial patterns for various indicator variables. Comparison thus involves three dimensions: space, time and indicators. Only cost-benefit analysis includes an explicit standard procedure to integrate both the different types of effect, since only those in monetary terms are considered, and the temporal pattern for each type of effect, namely via discounting. In multicriteria analysis the temporal or spatial pattern of effects can be included by specifying evaluation criteria that reflect the different spans of the effects. Via this route short- and long-term effects can be included in a single effects table and decision rule. Not all multicriteria methods are suitable for dealing with spatial and temporal issues (see Nijkamp *et al.*, 1991). Dynamic indicators can be aggregated in various ways. One approach is to discount each indicator pattern according to a specific procedure; even when only monetized values are included; varying discount rates can be used for separate variables, each measured in monetary terms (which is done in the

Table 8.1: Structure of a multicriteria decision problem

1. Problem definition
2. Description of the alternatives
2a) Continuous 2b) Discrete
3. Definition of criteria
3a) Latent 3b) Observable
4. Analysis of the impact of alternatives
4a) Qualitative information 4b) Quantitative information
5. Assessment of policy priorities
5a) Qualitative 5b) Quantitative
6. Selection of the alternatives
7. Presentation of the results
7a) Numerical 7b) Visual

Krutilla-Fisher algorithm). Alternatively, one may regard each data point, i.e., for a given indicator and point in time or space, as a separate attribute in the multicriteria evaluation procedure, and assign weights to each of these. Again, this is also possible when dealing with monetized values only. This approach will not be very attractive in general, since it requires the choice of too many weights, especially when temporal and spatial dimensions are considered simultaneously. Other approaches can focus on average values over time or space for indicators, or on final values. It should be noted that any weighting or discounting should take account of both temporal and spatial discounting and uncertainty discounting aspects. Finally, uncertainty, which is often crucial in ecological economics, requires a special treatment in multicriteria analysis. See, for a more extensive discussion of this and for empirical illustrations, Munda (1995). We will offer now some examples of applied multicriteria analysis.

8.3.1 An example of coastal habitats

An early example of a general ecological framework for conservation evaluation is the multiattribute ecological framework proposed by Randwell to evaluate coastal habitats (Randwell 1969). This method combines eight criteria into a single score, the Comparative Biological Value Index (CBVI). Each of these criteria is rated according the scale as described in Table 8.2. The final score is obtained by summing up the scores for all eight criteria:

$$CBVI = Ph + O + D + G + S + P + E + C$$

Table 8.2: Rating of the criteria used by Randwell (coastal habitats)

Criteria	Type		Rating
Physicochemical features (Ph)	High specialty		3
	Some special features		2
	Type example		2
Optimum populations (O)	Best populations of one or more local species		4
	Large populations of local species		3
	Large populations of common species and small populations of local species		2
Diversity (D)	Representative populations		1
	Outstanding populations		3
	High diversity		2
	Species ranges small		1
Geographic units (G)	Many species at limit		3
	Some species at limit		2
	Few species or no species at limit		1
Size (S)	Mud-flats (ha)	Cliffs (km)	5
	> 4,000	> 80	4
	1,600–3,999	40–79	3
	800–1,599	24–39	2
	400–799	8–23	1
	< 400	<8	
Purity (P)	Little disturbance		3
	Moderate disturbance		2
	Much ground disturbed or polluted		1
Education and research use (E)	Much used		3
	Some use		2
	Potential use		1
Combinatory value (C)	Adjacent to another habitat of likely national value		4
	Adjacent to another habitat of likely regional value		3
	Adjacent to another coast habitat site not spoilt by development		1
	Surrounded by developed coastline		0

Source: Randwell (1969).

The maximum value is 26 and the minimum value is seven. The higher the value, the more site protection is required. Since Randwell, the use of indices is a popular practice in ecological valuation and management (see Spellerberg 1992). However, this valuation approach relies on input criteria that require some subjective valuation. In fact, the group of experts do not only need to agree in fixing a list of criteria and attributes but also need to agree in assigning a weight for each of them. For example, any two different attributes with equal ratings of one can have a lower impact in the CBVI if an attribute is subjectively assigned a lower weight. Table 8.3 shows an example of the impact of manipulation of the weight of the attributes.

Table 8.3: An example of the manipulation of scores used to evaluate physical diversity

Criteria: Physical diversity	Score	Weight	Weight score	Maximum score	Maximum weighted score
Substrates	5	4	20	5	20
Fluvial features	3	5	15	5	25
Structure of aquatic vegetation	4	1	4	5	5
Total			39		50
Physical diversity index = 39/50 = 78%					

Source: Boon *et al.* (1997).

As we can see, the example refers to the evaluation of physical diversity, bearing in mind three attributes: substrates, fluvial features, and the structure of the aquatic vegetation. Each attribute is associated with a score (or rating) and a weight, which is the outcome of a sub-objective measurement process (for example a weight equal to one is given to the attribute with the lowest rank). If we combine both the score and weight of each attribute, we are able to compute the weight score (see column 'weight score'). Furthermore, if we combine the maximum score permitted for each attribute with its respective weight, we are to compute the maximum weight score (see the last column). The final index is expressed as a percentage of the maximum weighted score for that criterion, given the number of attributes for which the data were available, in this case 0.78 = (39 × 50) / 100.

8.3.2 Examples for the Netherlands

Ten Brink and Hosper (1989) have developed a general MCA valuation tool for the description and evaluation of ecosystems, i.e., an ecosystem classification method, known by the Dutch acronym AMOEBE (Algemene Methode voor OEcosysteembeschrijving en Beoordeling). The method was originally used to assess the quality of aquatic ecosystems by comparing the presence of selected species with their presence in a benchmark situation from 1930. The selected species, which ten Brink and Hosper called 'target variables', were selected on the basis of: (1) their representativeness (i.e., do they represent a healthy aquatic ecosystem); (2) their flexibility (i.e., can they be influenced by human interventions) and (3) their measurability and data availability (i.e., are they easy to measure and are databases available).

In 1995, the Dutch Centre for Agriculture and Environment developed the nature measurement method to assess the natural values of agricultural areas. These were measured in terms of species abundance and its deviation from its own diversity potential. Species were selected that are present on agricultural plots, easy to recognize and that represent natural quality (Buys 1995). A similar formula was developed by the Foundation for Spatial Economics of the University of Groningen to assess the costs and benefits of the National

Ecological Network (Sijtsma and Strijker 1995). The costs were valued entirely in monetary terms and the benefits mostly in ecological terms. The National Ecological Network is based on the identification of 'nature target types' (i.e., pre-defined types of nature such as the European CORINE network) that are assumed to provide the habitat mosaic for 'target species'. To this purpose, digital thematic maps are generated as geographic information systems with land cover habitat types. Target species are, in turn, classified and selected on the basis of national and international rarity. Zurlini *et al.* (2000) have recently applied this valuation method to create a map of Italian nature.

In 1996, the Center for Environmental Studies in Leiden developed an ecological effect measurement method to value the effects of housing development projects on nature and landscape. The method consists of three steps: a description of the reference situation and human intervention measures; a determination of the effects on nature; and an aggregation of effects (Cuperus and Canters 1995). The main ecosystem's biotic and abiotic characteristics were used for valuation, including spatial diversity (e.g., variables on soil and vegetation structures), abiotic functioning (e.g., variables on temperature, sediments, water and soil), and fauna and flora communities as well as their relations with the surroundings (e.g., variables on hydrologic, geomorphologic and biomass relations). Since this valuation method takes into account both biotic and abiotic diversity as assessed by means of the deviation from a reference situation, it can easily be transferred to the policy arena and used, for example, to determine compensation measures in the case of damage to existing natural areas.

The multicriteria valuation method was designed to correspond with the information needs of the decision-makers. The Dutch Land Water Impulse, using the results of other available methods as inputs (Oppers and Ruijgrok 1997; Ruijgrok *et al.* 1999), originally developed this method to support management decisions concerning the conservation of nature along the Dutch coast – see Ruijgrok (1999). Unlike the other methods, this one is specifically aimed at indicating whether one ecosystem is more valuable that another. This valuation method combines the use of a set ecological evaluation criterion covering the ecologically important processes and pattern criteria without introducing overlap. Naturalness and replaceability are process criteria, whereas diversity and rarity are pattern criteria (see Table 8.4). The six criteria are put into a multicriteria framework, i.e., a matrix with the management alternatives in its columns and criteria scores in its rows. Total scores are calculated by means of a weighted summation. More recently, the Dutch Environmental Planning Bureau in Bilthoven developed the ecological capital index with the objective of assessing the state of both natural and cultural ecosystems in relation to human activities. This ecological capital index is calculated by combining the ecosystem's quantity, measured in terms of the deviation of ecosystem's present area from its area in a reference situation, with the ecosystem's quality,

Table 8.4: Ecological valuation criteria

Characteristics	Criteria	Measurement
Pattern (space)	Biotic rarity	International prevalence
	Abiotic rarity	Number of ha of nature target types
	Biotic diversity	Number of species
	Abiotic diversity	Number of soil and water characteristics
Process (time)	Naturalness	Implicitly accounting in the absence of human influence
	Replaceability	Regeneration time after destruction

Source: Oppers and Ruijgrok (1997).

measured in terms of the deviation of ecosystem's present species diversity from its diversity in a reference situation (RIZA 1999). The international application of the ecological capital index respects the recommendations for a core set of indicators, such as those proposed by the Convention on Biological Diversity (UNEP 1997), and is therefore compatible with the International Union for Nature Conservation (UNEP)'s classification of ecosystems on the basis of the degree of human influence. UNEP distinguishes four types of ecosystems. These are natural, adapted, cultivated, built and deteriorated ecosystems (IUCN 1991).

In conclusion, multicriteria analysis has a great potential in applied research for biodiversity. It derives its strength mainly from the systematics in organizing the relevant policy information and from the ability to include a multidimensional data with different measurement levels. It may be regarded as a strong communication tool in biodiversity policy.

Table 2.4. Ecological valuation of areas

Characteristics	Criteria	Measurement
Rarity	Biodiversity	International prevalence
	Biodiversity	Number of (endangered) species
	Naturalness	
	Biodiversity	Number of species
	Biodiversity	Number of (endangered) species
	Naturalness	Ecosystem functioning in the absence of human influence
Process		
(Recovery)	Replaceability	Susceptibility and time required for recovery

Source: Crone and Ruijgrok 1997.

measured in terms of the deviation of ecosystem's present species diversity from its diversity in a reference situation (RIZA 1999). The international application of the ecological method must require the recommendations for a core set of indicators, such as those proposed by the Convention on Biological Diversity (CBD, 1997), and is therefore compatible with the International Union for Nature Conservation (IUCN)'s classification of ecosystems on the basis of the degree of human influence. IUCN distinguishes four types of ecosystems. These are natural, modified, cultivated, built and constructed ecosystems (IUCN 1991).

In conclusion, multicriteria analysis has a great potential in applied research for biodiversity. It derives its strength mainly from the systematics in organising the relevant policy information and from the ability to include a multidimensional data with different measurement levels. It may be regarded as a strong communication tool in biodiversity policy.

[illegible] method to support management decisions concerning the conservation [illegible] (Crone and Ruijgrok 1997). Unlike the other methods, [illegible] indicating whether one ecosystem is more valuable than another. The following method [illegible] the use of a [illegible] ecological [illegible] naturalness [illegible] (see Table 2.4). [illegible] by means of a weighted summation. More recently, the Dutch Environmental Planning Bureau [illegible] developed the ecological capital index [illegible] calculated by combining the ecosystem's quantity [illegible] with the ecosystem's quality.

Chapter 9

Research Synthesis and Value Transfer

9.1 Introduction

The ecological economics of biodiversity finds its focal point at the crossroads of natural and human values of ecosystems. In addition, to a resolution of methodological complexities there is also the need to look for general lessons and findings from past applied research. The large number of applied economic valuation studies currently available has induced the search for commonalities in different empirical investigations and has also led to the current popularity of meta-analysis and value-transfer. In particular, in recent years we have seen a rising number of publications on the economic aspects of biodiversity, both theoretical and empirical. This prompts the intriguing question of whether a more general conclusion might be inferred from a set of specific empirical investigations on closely related research themes or issues. Meta-analysis has originally been developed in the context of the natural sciences (in particular, in medical sciences) as a statistical tool for use in comparative studies and for creating synthetic knowledge from controlled experimentation studies (see Florax *et al.* 2002 for a detailed review). More recently, meta-analysis has been diffused to the social sciences, including psychology, sociology and economics. This has generated a new stream of quantitative research seeking a synthesis of scientific results based on statistical applied analysis. The application of conventional statistical methods in meta-analysis is particularly appropriate when the case focuses on identifying trends and relationships between the variables being analyzed. The methods are also appropriate for the construction of synthetic indicators and the determination of parameters or other common elements that can be described in quantitative terms (more details are to be found in Hedges and Olkin 1985; Hunter and Schmidt 1990; Wolf 1986).

Meta-analysis refers to the use of quantitative methods, mainly statistical, applied for the comparison or synthesis of outcomes from a set of empirical investigations on a common, or largely similar, issue (Stanley and Jarrel 1989; Cooper and Hedges 1994). In contrast, value transfer aims to develop a quantitative framework for the transferability of value (or benefit) estimates for policy decisions. Currently, many efforts are undertaken to carry out comparative studies using a framework offered by meta-analysis and value transfer (for a recent survey, see van den Bergh *et al.* 1997). In this context, applied environmental economic research has in recent years proposed to develop a test on value transfer by conducting two parallel case studies with the aim of deriving non-market environmental values and comparing them with the obtained results (see Bergland *et al.* 1995; Kirchhoff *et al.* 1997; Kirchhoff 2000 and Bateman *et al.* 1995). Besides the case study approach used to obtain the required values for value transfer purposes, statistical methods based on meta-analysis can be used to obtain quasi-estimations of non-market values (see van den Bergh and Button 1999). Thus comparative studies seem to cover new ground in economics, in general, and in environmental economics, in particular.

Meta-analysis has many advantages in applied quantitative research. It avoids the need to develop a costly and new methodological basis for new case studies, as far as they address similar issues as past studies. It allows for the statistical identification of major driving factors in causal relationships and can also act as a robustness check on the existing studies. In the context of value transfer, such research synthesis experiments also offer operational frameworks for making conditional forecasts.

9.2 Meta-analytical methods

9.2.1 Statistical techniques

Meta-analytical methods have gained much popularity, not only in the research synthesis of previous case study surveys, but also as a tool for comparative case study research and benefit-transfer in case of uncertain outcomes in different situations. In recent years meta-analysis has been applied in environmental economics, particularly in studies that have performed monetary valuation. Table 9.1 gives a selected list of representative studies. Meta-analysis of urban environmental issues has focused mainly on hedonic property value models. With regard to ecosystems and biodiversity, method comparisons have received attention, notably of contingent valuation and revealed preference.

Carson *et al.* (1996) undertook 83 studies, which included 616 comparisons of contingent valuation (CV) and revealed preference (RP) estimates. The latter were based on applications of travel cost and hedonic pricing techniques, expenditure and household production function

Table 9.1: Applications of meta-analysis

Subject area	Studies
Urban pollution valuation	Smith (1989), Smith and Huang (1993), Smith and Huang (1995), Schwartz (1994), van den Bergh *et al.* (1997)
Recreation benefits	Smith and Kaoru (1990a), Walsh *et al.* (1989a, b)
Recreational fishing	Sturtevant *et al.* (1995)
Water quality	Magnussen (1993)
Valuation of life estimates	van den Bergh *et al.* (1997)
Contingent valuation vs. revealed preference	Carson *et al.* (1996)
Wetlands valuation	Brouwer *et al.* (1999), Woodward and Wui (2001)
Noise nuisance	Nelson (1980), Button (1995), Schipper (1997)
Travel congestion	Waters (1993)
Visibility improvement	Smith and Osborne (1996)
Transport issues	Button (1995), Button and Kerr (1996)
Multiplier effects of tourism	Nijkamp and Baaijens (2001)
Price elasticity of demand and travel cost	Smith and Kaoru (1990b)
Price elasticity of gasoline demand	Espey (1996)
Valuing morbidity	Johnson, Fries and Banzhaf (1996)

Source: van den Bergh and Button (1999, Table 55.1, p. 800).

models, and simulated or actual market creation. The estimates cover a variety of environmental issues that fall into the class of quasi-public goods, and include recreation benefits, water quality improvements, fishing, mining impacts, forest harvesting, and preservation. These were then classified into three broad classes, namely recreation, environmental amenities (air and water quality) and health risks. It was found that CV estimates are generally smaller than their RP counterparts, even if only slightly. Meta-analysis has been applied to ecosystem valuation studies, notably to improve our understanding of the links between functions and values. Individual ecosystem valuation studies provide limited insight into this, as they do not contain sufficient variation in quantitative information about ecosystem processes as well as functions. A meta-analysis, instead, can lead to insight about which values and functions are relatively important.

Brouwer *et al.* (1999) performed a meta-analysis using 30 contingent valuation studies of wetlands. They added information to the quantitative output of the primary studies, namely by identifying wetland type, main functions, country, type of value estimated, and selected characteristics of the contingent valuation survey design. Their analysis suggests that the average willingness to pay was highest for flood control, followed by supply of water, water quality, and the provision and maintenance of biodiversity.

The conventional statistical methods have often been applied in order to compare outcomes from different methods on a given issue (van den Bergh *et al.* 1997; Florax *et al.* 2002). Moreover, different outcomes from various studies can be explained by the differences in the formulation of the research, the size and type of data analysed, statistical methods applied, and by temporal and geographical characteristics of the studies. The use of quantitative and qualitative multicriteria techniques is of great importance when the objective of the synthesis is the qualitative comparison of studies and when the result of studies can be interpreted as 'criteria'. The application of these methods is especially important when the studies do not provide one simple value per indicator or when the analysis focuses on a number of indicators. In principle, meta-analytical methods are able to encapsulate these components.

9.2.2 Other techniques

Meta-analysis does not consist solely of the application of purely quantitative-statistical techniques to synthesize research outcomes. The format of study findings and the objective of their synthesis in many areas of the social sciences, including economics, do not exclusively stem from the experimental investigation of phenomena. Therefore, statistical meta-analytical methods and statistical inference, as they are usually used in scientific research, do not reveal themselves as the only or the most suitable techniques for synthesis (van den Bergh *et al.* 1997). As a result, over the past decades several new non-statistical methods that synthesize study findings from many areas of the social sciences have become available. The alternative non-statistical or non-parametric statistical techniques used for synthesis can be classified into main categories, in particular: (1) rough set analysis; (2) fuzzy set analysis; and (3) content analysis. These tools appear to show much promise for drawing quantitative inferences on the basis of even a collection of qualitative study findings (Hogenraad 1989). This section presents a concise introduction and discussion of these meta-analytical methods.

9.2.2.1 Rough set analysis

Rough set analysis – developed by Pawlak (1982) – is the most suitable technique for synthesis when the studies are to be grouped and classified according to numerical characteristics that are imprecisely measured. This analysis originates from artificial intelligence and aims to pinpoint data regularities that are not immediately evident; it searches for the possible existence of the principle of causality among data sets and attempts to eliminate irrelevant information. An important feature of this synthesis technique is that it does not necessarily require numerical information about the data being used, provided it is classified in distinct groups. In this way it is able to synthesize a mixture of classified qualitative and quantitative data as well as to combine study findings that are subject to inconsistencies and inaccuracy (Pawlak 1982; Slowinski and Stefanowski 1989). In fact, rough set theory also has the advantage that it is able to create a ranking of actions in multiattribute decision support processes (van den Bergh *et al.* 1997). Given its features, it is clear that rough set analysis is a readily applicable synthesis technique for decision-making processes that deal with a large input of similar study findings. Several software packages are at present available.

9.2.2.2 Fuzzy set analysis

Fuzzy set analysis is applicable for the analysis of the same category of issues as rough set analysis. However, it is the most suitable technique for synthesis whenever study findings contain a clear component of linguistic uncertainty in terms of imprecise measurement (see, for example, Kacprzyk 1978 and Munda *et al.* 1993). As Dubois and Prada (1992) have made clear, the distinction between fuzzy set analysis and rough set analysis is more subtle than is commonly expected. For example, instead of the discrete classes used in rough set analysis, fuzzy set analysis employs a continuous classification scale. Fuzzy sets refer to linguistically defined variables that do not have an unambiguous measurement scale. Fuzzy variables can be categorized into classes for which the boundaries are weakly demarcated, so that variables can belong to these classes to some degree. A good example of the use of fuzzy set theory in environmental quality valuation can be found in Munda (1995), who also developed the NAIADE software package.

9.2.2.3 Content analysis

Content analysis can be defined in the following way: content analysis is a method for making inferences by identifying characteristics of text messages in a systematic way in order to convert the text message into distinct classes that can be studied with the use of quantitative methods. Therefore, this synthesis technique is able to consider simultaneously the dynamic context of cultural, social, economic and political aspects. Content analysis is able to synthesize all kinds of verbal messages and texts by means of quantitative methods, inducing a mapping of specific types of words in a text into fewer content categories. Besides coding, sophisticated computer software can be used, such as the LISREL econometric package (see Weber, 1983). It is clear that a collection of published study findings from previously undertaken case studies rather than a single text may serve as the basis for content analysis. Many interesting applications can be thought of: newspapers, Internet sites, and scientific magazines can be studied. Even when we limit ourselves to the social sciences, a wide range of application possibilities can be observed. According to Weber (1983, p. 128), in general, 'social scientists will be able to use computer-aided content analysis with greater confidence to address a wide variety of theoretical problems involving the relationships among cultural, social, economic and political change. As Weber (1983) points out, content analysis is not only capable of considering mutual socio-economic relationships but also their dynamic aspects, which are highly useful in economic research. In particular, content analysis can be used to generate a flow of benefits over time and to compute net present value, which can serve as a value relating to a particular scenario of ecosystem change or management. Studies that consider these features of content analysis are, for example, Namenwirth (1969) and Rosengren (1981). These show the existence of a strong relationship over time between political statements and the state of the economy. It is clear that content analysis is also a powerful tool to extract quantitative or code information from qualitative data input.

9.3 Value transfer

A procedure that makes the use of meta-analysis is benefit or value transfer. This refers to the transfer of research findings. In the context of ecosystem and biodiversity valuation, the general idea is to explore the use of previous and original valuation studies ('study site') and transfer their estimates' values to the site where the new value estimate is needed ('policy site') – see Navrud and Bergland (2001); and Brouwer *et al.* (1999). Value transfer brings up an important research question: what lessons can be drawn from a comparative analysis of monetary estimates derived from earlier empirical studies for an additional similar case not included in the meta-sample? The solution is to perform essentially a meta-analysis, and to use the estimation results to predict for the new or yet unknown empirical case. In economics, value transfer can be explored across different sites – spatial value transfer – or, for one specific site or valuation object over time – temporal valuation transfer.

A major advantage of value transfer for policy guidance is that it ensures more comparability and consistency across different evaluation studies. It may also be helpful in an initial screening of a large number of public projects that cannot be investigated in full detail. Furthermore, the outcome of a previous study may be used as a benchmark against which results of studies can be evaluated. But the most important advantage of benefit transfer is that it is a cost-effective way to make quantitative statements about phenomena that have not been subject to previous analysis (see Johnson and Button 1997).

In empirical research, two value transfer approaches are available: unit value transfer and function value transfer. Both can be based on the principles of meta-analysis. The former transfers mean monetary value estimates, for example mean willingness to pay, directly from the study site to the policy site, with possible income adjustments. The latter, instead of transferring individual willingness to pay estimates, explores the use of more information, such as characteristics of the object of valuation and the subject who performs the valuation exercise, and can generate a benefit function that allows 'prediction' for the policy site. Examples of these approaches can be found in income elasticity studies, savings rate studies, consumer's surplus studies, accessibility studies, value of time studies and evaluation studies on environmental decay. The implicit assumption is, then, that the degree of variation in estimated parameters is sufficiently small to be able to deploy valuations from the original data in a given case study to assess corresponding parameters from other similar cases, usually in different contextual settings. Clearly, the more uniform the set of previous studies, the more likely the validity of the above implicit assumption (see Smith and Kaoru 1990a; Smith 1992).

In conclusion, next to the quantitative statistical techniques, such as standard meta-regression analysis, there is a variety of complementary methods which also offer a great potential in research synthesis in ecological economics of biodiversity. In Section 9.4 we will give some empirical illustrations.

9.4 A comparative study of biodiversity values

9.4.1 Background

Meta-analysis aims to offer a framework for quantitative research synthesis. But it may also serve as a quantitative framework for the comparative analysis of previously undertaken studies. In the present section, a meta-analysis study will be presented based on a data set provided by the Dutch National Institute of Public Health and the Environment (RIVM) (ten Brink *et al.* 2000). It contains a list of empirical case studies relevant to the valuation of different aspects of biodiversity and different kinds of habitat, ranging from wildlife and endangered species preservation to the protection of national parks and nature areas – Table 9.2 provides a full list of the case studies. As we can see, the database includes not only data about the type of environmental quality under assessment, but also information about the authors of the study, the country (all studies are localized in Northern Europe), the particular valuation method deployed, and the year in which the study took place. Furthermore, the database shows the use of various economic valuation methods as well as different measurement units or scales in which the monetary benefits are expressed. These, in turn, complicate the derivation of an aggregate, standardized measure of the monetary value, which would allow us to calculate the total annual environmental benefit. In order to shed light on this type of monetary value estimate, a set of 'compatible studies' was used, reflecting a 'similar method of estimation' (ten Brink *et al.* 2000, p. 25). Table 9.3 presents the standardized willingness to pay measure per person per year for biodiversity preservation and different kinds of habitat. The willingness to pay (WTP) figures have been converted to € 1997.

Table 9.2: Biodiversity and habitat values

Type of good Study [a]	Range of the monetary value estimate	Country	Method	Year of study
Biodiversity				
Spash and Hanley, 1995	£8.01–£15.42 once-off payment per person	UK	CV-OE	1993
Spash and Hanley, 1996	£46.99–£62.26 per household per year	UK	CV-OE	1993
Navrud, 1992	NOK 194 annual payment per year	Norway	CV-DC	1991
Holm-Muller *et al.*, 1991	DM 16.1 per month	Germany	CV	1990*
Macmillan *et al.*, 1995	£75 per household	UK	CV-OE	1994
Macmillan *et al.*, 1995	£308 per household	UK	CV-DC	1994
Wildlife				
Willis *et al.*, 1996	£13.83– £61.74 per household per year	UK	CV-PC	1994
Willis, 1990	£0.98–£3.12 per visitor per year	UK	CV-OE	1986
Willis, 1990	£13–£65 ha per year	UK	TCM	1986
Willis and Benson, 1988	£25 ha/year per person	UK	CV-OE	1985
Willis and Benson, 1988	£0.6–£1.7 visit per person	UK	TCM	1985
Willis and Benson, 1988	£1.02–£2.3 visit per person (full travel cost)	UK	TCM	1986
Hanley, 1989	£1.18–£2.53 per adult	UK	CV	1988
Harley and Hanley, 1989	£1.99–£2.60 per person	UK	ZTCM	1988
Hanley, 1991	£16.8 once-off payment per person	UK	CV	1990*
National parks and nature reserves				
Szereny, 1997	£6 per visitor	Hungary	CV-PC	1996
Willis and Garrod 1991	£24 per household	UK	CV	1990
Willis, 1990	£4.54 per person	UK	CV	1985
Willis, 1990	£0.82 per person	UK	CV	1986
Watercourses				
Hervik *et al.*, 1987	\$50–\$100 per person per year	Norway	CV	1990
Tapsell *et al.*, 1991	£0.75–£0.95 per adult visitor	UK	CV	1991
Willis and Garrod, 1991	£0.36 per visitor	UK	CV	1989
Coker *et al.*, 1989	£13.90–£16.20 per household per year	UK	CV-OE	1988
Green *et al.*, 1990	£13.59–£5.56 per person per year	UK	CV	1987
Green and Tunstall, 1990	£546–£582 once-off payment per household	UK	CV-OE	1987
Green and Tunstall, 1991	£12.08 per person per year	UK	CV	1987
Garrod and Willis, 1994	3-5% increase in property sale price	UK	HPM	1989
Willis *et al.*, 1990	£9.2–£11.2 per person per visit	UK	ITCM	1988
Garrod and Willis, 1991	4.9% increase in property sale price	UK	HPM	1990*
Willis and Garrod, 1990	£0.51–£3 per visit	UK	TCM	1989*
Landscape				
Pruckner, 1995	ATS 9.2 per visitor per day	Austria	CV-OE	1991
Drake, 1991	SEK 750 per person per year	Sweden	CV-PC	1991
Bullock and Kay, 1997	£49–£55 per household per year	UK	CV-DC	1994
Spaninks, 1993	NLG 55 per household per year	Netherlands	CV-PC	1993
Brouwer, 1995	NLG 80 per household per year	Netherlands	CV-OE	1994

Source: RIVM (2000).

Note: (a) as cited in RIVM (2000); CV = contingent valuation; CR = contingent ranking; OE = open ended; PC = payment card; DC = dichotomous choice; TCM = travel cost method; ZTCM = zone travel cost method; -- = valuation method unspecified, and * = base year unspecified and assumed to be the year before presentation of the paper.

Table 9.2: Biodiversity and habitat values (cont.)

Type of good Study [a]	Range of the monetary value estimate	Country	Method	Year of study
Endangered species				
Strand, 1981	NOK 1,700–NOK 2,750 per person per year	Norway	CV	1991
Dahle *et al.*, 1987	$15 per person per year	Norway	CV	1990
Johansson, 1987	$7 per person per year	Sweden	CV	1990
Johansson, 1989	SEK 85 per person per year	Sweden	CV	1991
Garrod and Willis, 1994	£2.94 per year	UK	CV	1993*
Fredman, 1994	SEK 406 per person	Sweden	CV-DC	1993
Wetlands				
Gren *et al.*, 1994	£67 per household per year	UK	CV-OE	1991
Gren *et al.*, 1995	£75 per household per year	UK	CV-IB	1991
Kosz, 1996	ATS 329.25 per Austrian per year	Austria	CV-OE	1993
Bateman *et al.*, 1995	£21.75 per household per year	UK	CV-OE	1991
Bateman *et al.*, 1995	£76.74 per visitor per year	UK	CV-OE	1991
Bateman *et al.*, 1995	£83.67 per visitor per year	UK	CV-IB	1991
Woodlands				
Bateman *et al.*, 1996	£9.94 per household per year	UK	CV-OE	1991
Garrod and Willis, 1998	£18.5–£20.7 per household per year	UK	CR	1995
Van der Linden *et al.*, 1987	NLG 22.83 per household per month	Netherlands	--	1987
Kristrom, 1990	SEK 95 per person per year	Sweden	--	1991
Johansson and Kristrom, 1988	SEK 1,014 per household per year	Sweden	CV-OE	1988
Kristrom, 1990	$3–$4 per person per year	Sweden	CV-OE	1988
Johansson and Zavisic, 1989	$5–$8 per person per year	Sweden	CV	1990
Hoen and Winther, 1991	$13–$18 per person per year	Norway	CV	1990
Bateman *et al.*, 1991	£1.21–£7.09 per person per year	UK	CV-OE	1991
Hanley, 1991	£9.73 per visitor per year	UK	CV	1990
Willis and Benson, 1988	£0.53 per visitor	UK	CV	1988
Willis *et al.*, 1988	£0.33 per visitor	UK	CV	1987
Willis and Benson, 1989	£1.3–£3.3 per visit	UK	ZTCM	1988*
Hanley, 1989	£1.25 per visit	UK	CV-PC	1987
Hanley, 1989	£15.13 per visitor per year	UK	TCM	1988*
Hanley and Common, 1987	£1 per visitor	UK	CV	1986*
Hanley and Common, 1987	£14.6–£24.5 per visitor per year	UK	TCM	1986*
Garrod and Willis, 1991	7.1% increase in property sale price	UK	HPM	1990
Garrod and Willis, 1991	£43 increase in property sale price	UK	HPM	1988
Garrod and Willis, 1991	£0.06–£0.96 per visitor	UK	TCM	1988
Garrod and Willis, 1991	£0.43–£0.72 per person	UK	CV	1988
Willis, 1991	£1.95 per visitor	UK	TCM	1988
Willis, 1991	£0.53 per visitor	UK	CV-OE	1988
Bateman and Langford, 1995	£12.55 per year	UK	CV	1994*
Bateman *et al.*, 1995	£3.51 per year	UK	CV	1994*
Bateman *et al.*, 1995	£1.82–£2.78 per visit	UK	CV-OE	1990
Hanley *et al.*, 1991	£20.6–£30.59 per person	UK	CV	1990*
Hanley and Ruffell, 1991	£0.33–£0.93 per visitor per visit	UK	CV-OE	1991

Source: RIVM (2000)

Note: (a) as cited in RIVM (2000); CV = contingent valuation; CR = contingent ranking; OE = open ended; PC = payment card; DC = dichotomous choice; TCM = travel cost method; ZTCM = zone travel cost method; -- = valuation method unspecified, and * = base year unspecified and assumed to be the year before presentation of the paper.

The data basis of Table 9.3 contains both numerical information (such as willingness to pay) and alpha-numerical, linguistic and categorical information (e.g., country, year of study, etc.). This makes the application of standard statistical tools rather problematic. Nevertheless, even a linguistic information base may incorporate a hidden structure in terms of associations between patterns, or the frequency of occurrence of a given phenomenon (or qualitative characteristic or parameter). In this context, we may resort to qualitative pattern recognition methods, elucidated inter alia in the artificial intelligence literature. There is a wide variety of such methods, such as computational neural networks, genetic algorithms, fuzzy and rough set methods, decision tree induction methods, etc. An interesting recently developed algorithm in this framework is the a priori algorithm, which is able to identify association rules among qualitative data (Agrawal *et al.* 1996). This meta-analytical method is concisely described in Section 9.4.2 and Section 9.4.3 presents the quantitative results.

9.4.2 Meta-analytical method used

The a priori algorithm belongs to the rough set methods. It seeks to identify patterns among frequencies of qualitatively described phenomena by using complex learning rules. Such rules show under which conditions a certain association rule (e.g., a combination of A and B leads to C) may occur. It seeks to identify patterns among frequencies of qualitatively described phenomena by using complex learning rules. Such rules show under which conditions a certain association rule (e.g., a combination of A and B leads to C) may occur. This method is particularly relevant for our study, as the categorical or qualitative information may be summarized in Boolean algebra. In such a context, a relation R contains n–tuples over a set of Boolean attributes $A_1, A_2, \ldots A_n$. Let $I = \{ A_{i1}, A_{i2}, \ldots A_{in} \}$ and $J = \{ A_{j1}, A_{j2}, \ldots A_{jn} \}$ be two sets of attributes. Then $I \Rightarrow J$ is an *association rule*, if the following two conditions are satisfied: the *support* for the set $I \cup J$ appears in at least an s–fraction of the tuples; and a *confidence* amongst the tuples shows up if at least a c–fraction also has J appearing in them. The analysis aims to identify all valid association rules within the database for the valuation of different biodiversity and different kinds of habitats. The first step in the a priori analysis is to split the database into two distinct sets of variables: viz. on the basis of (1) valuation method, and (2) the good under valuation – see the left-hand side (LHS) of Table 9.4. The second set is characterized by (3) the price variable – see the right hand side (RHS) of Table 9.4. In addition, Table 9.4 contains information about the number of studies that support an association rule to be identified and its relative share in terms of frequency, which incorporates the weight of the association rule on the database. Finally, Table 9.4 contains information concerning the strength of the association rule within the database.

Table 9.3: Mean willingness to pay for biodiversity and habitat services

Type of good	WTP per person per year (in €) Unweighted	Adjusted to EU	Countries of the study
Biodiversity preservation	31.1	28.66	UK, Norway, Germany
Wildlife preservation	1.9	1.8	UK
National parks and nature reserves	9.4	8.7	UK, Hungary
Wetlands	38.0	35.0	UK, Austria
Watercourses	31.1	27.2	UK, Norway
Landscape	62.5	57.5	UK, Netherlands, Austria, Sweden
Endangered species protection	128.6	120.9	UK, Sweden, Norway
Woodlands	20.1	18.8	UK, Netherlands, Sweden, Norway

9.4.3 Results

The application of the a priori algorithm leads to interesting research findings. Table 9.4 shows that that there are 18 possible association rules, which correspond to the most frequently appearing regularities in the database. One particularly interesting association rule that emerges from the database links (2) the good under valuation to (3) the price, which is interpreted in terms of WTP. For example, if the good under consideration is 'woodland habitat', the corresponding association rule (see first row) shows that the woodland monetary valuation falls between €9.4 and €20.1. This price range is supported by 30 cases, which corresponds to 38 per cent of all case studies in the database. The empirical magnitude of this association rule appears to be very high: it corresponds to a confidence level of 100 per cent (see the last column). In other words, all the 'woodland habitat' valuation cases support this association rule. If the good under consideration is 'biodiversity', this price-good association rule (see row 6) shows us that the respective price ranges between €20.1 and €31.1, which is true for all biodiversity case studies presented in the database. However, the empirical significance of this rule is rather low, since it is only valid for 7.6 per cent of the case studies in the database, i.e. only 7.6 per cent of the sample of case studies were concerned with biodiversity.

Another association rule that emerges from the database links (a) the method of valuation to (c) the price. This rule shows (see row 9) that if the valuation method under consideration is contingent valuation the respective monetary estimates range between €9.4 and €20.1. This is true for 13 of all the 27 case studies that used CV as the selected valuation method. Therefore, the empirical magnitude of this association rule corresponds to a confidence level with a strength of 48.1 per cent. In other words, about half of the CV valuation cases support this price association rule, which corresponds to 16.5 per cent of the database. A similar price association rule can be found for the TCM, i.e., the respective price estimates range between €9.4 and €20.1 (see row 12). However, despite the fact that this price rule appears to have

Table 9.4: Results for alternative association rules

Association rule Good, method (LHS) If:	Price in € (RHS) Then:	Share of studies (%)	Number of studies	Confidence level (%)
(1) Good = woodlands	9.4 < price ≤ 20.1	38.0	30	100.0
(2) Good = watercourses	20.1 < price ≤ 31.1	13.9	11	91.7
(3) Good = wildlife	price ≤ 9.4	11.4	9	100.0
(4) Good = endangered species	price > 38.0	8.9	7	100.0
(5) Good = wetlands	31.1 < price ≤ 38.0	7.6	6	100.0
(6) Good = biodiversity	20.1 < price ≤ 31.1	7.6	6	100.0
(7) Good = landscape	price > 38.0	6.3	5	100.0
(8) Good = national parks and nature reserves	price ≤ 9.4	5.1	4	100.0
(9) Method = CV	9.4 < price ≤ 20.1	16.5	13	48.1
(10) Method = CV-OE	31.1 < price ≤ 38.0	5.1	4	20.0
(11) Method = CV-OE	20.1 < price ≤ 31.1	6.3	5	25.0
(12) Method = TCM	9.4 < price ≤ 20.1	5.1	4	57.1
(13) Good = woodlands & method = CV	9.4 < price ≤ 20.1	16.5	13	100.0
(14) Good = woodlands & method = TCM	9.4 < price ≤ 20.1	5.1	4	100.0
(15) Good = woodlands & method = CV - OE	9.4 < price ≤ 20.1	8.9	7	100.0
(16) Good = wetlands & method = CV - OE	31.1 < price ≤ 38.0	5.1	4	100.0
(17) Good = watercourses & method = CV	20.1 < price ≤ 31.1	6.3	5	83.3
(18) Good = endangered & method = CV	price > 38.0	5.1	4	100.0

stronger empirical support because it corresponds to a higher confidence level with a strength of 57 per cent, this association rule is valid for only 5.1 per cent of the case studies registered in the database under consideration.

In addition, one can look at association rules that combine both the good and the method and link them to the price. As we can see from Table 9.4 such rules have a high strength, but must be interpreted carefully, since they appear to be associated with a very low level of support, since they apply only to a very limited number of case studies within the data set. For example, an association rule that is characterized by setting good = watercourses and method = CV, and the price to range between €20.1 and €31.1 appears to be valid for only 6.3 per cent of the case studies registered in the database. However, this association rule corresponds to a high level of strength, an 83.3 per cent confidence level.

To conclude, one can infer that the present database is somewhat too small for the identification of robust compound rules. It seems that the relatively aggregate and standardized valuation figures are too imprecise to support firm conclusions and thus too poor to be used for policy guidance. Nevertheless, the technical method deployed there indicates clearly that there is no more reliable quantitative information to be extracted from the present database. In general, the above described multidimensional classification methods are in principle able to identify relevant causal linkages and structures in a qualitative database.

PART V
Policy and Conclusions

Chapter 10
Biodiversity Policy

10.1 Introduction

Biodiversity comprises functions that affect the well-being of individuals and societies in all regions of our world. The mainly public good character of biodiversity, combined with the presence of many externalities, evidently gives rise to a market failure. In particular, market prices fail to capture the full range of biodiversity benefits to individuals and society. This contributes to the rapid depletion of biodiversity, leading to important welfare losses. Therefore, there is a clear scope for public biodiversity policy. A successful public policy design aimed at ensuring the conservation and sustainable use of the full range of biodiversity benefits implies the use of environmental instruments that: (1) protect biodiversity private values, such as benefits in terms of species and genetic diversity (see link 2 → 5 in Figure 5.1), through the provision of proper market incentives, such as taxes and charges or the assignment of well-defined property rights; and (2) protect biodiversity public values, such as benefits in terms of the knowledge of the continued existence of ecosystems diversity and bequest values related to maintaining them for the enjoyment of future generations (see link 3 in Figure 5.1), through the use of institutions and the creation of market mechanisms such as the provision of information. Therefore, biodiversity policy-related measures are often applied in a policy mix, combining standard-setting, direct and indirect market intervention and the provision of information. Table 10.1 presents a list of the strategies available to governments involved in biodiversity policy design. These are discussed in more detail in the subsequent sections.

Table 10.1: Strategies for government involvement in public biodiversity policy

Direct government interventions:

1. Price incentives: fees, charges, taxes and tradable permits.
2. Command-and-control strategy: quantity standards, technology regulation, access restrictions.
3. Assignment of property rights.

Provision of information:

4. Development of market mechanisms: certification, ecolabeling, and institutional building.

Source: OECD (1999).

10.2 Public policy strategies

10.2.1 Direct government intervention

One possible way of addressing market failures is through direct government intervention. This involves the use of policy instruments such as taxes, command-control policy, and the definition of property rights. The best-known price instrument is the optimal or Pigouvian tax, which restores a situation with biodiversity externalities to a social optimum. For example, in 1995 the Dutch government introduced a groundwater tax so as to minimize the desiccation effects associated with the excessive use of groundwater reserves, one of the most important causes of biodiversity loss in the country (Bellegem *et al.* 1999). However, the implementation of Pigouvian taxes for policy design is very difficult in view of the large degree of uncertainty associated with determining the social costs of biodiversity loss and the high financial expenses of the activities involved in translating such social costs into monetary terms. As a result, it is difficult to find other situations where policy instruments make the use of Pigouvian taxes to internalize the full non-market benefits of biodiversity.

Alternatively, the government can impose strict command-control policies. This means that the government directly dictates clear quantity targets, i.e., quantity standards, which have to be followed by producers and consumers. The government may have to set up a regulatory body, which monitors whether the restriction is being complied with by the firms and which enforces it by punishing violators. An example is to set a limit on the number of daily visitors to nature areas, such as sensitive wilderness areas, or on the number of animal species that can be caught by hunters or fishers. Adopting such quantity standards is especially attractive from the perspective of policy effectiveness. However, command-control policy generally implies embracing high monitoring and enforcement costs. Moreover, uniform control does not sufficiently address the heterogeneity of agents, and thus misses out on potential efficiency gains.

Third, the government can opt for the provision and enforcement of well-defined property rights. This instrument is particularly efficient in addressing the market price internalization of the private values of biological resources. An example of this type of public policy is the assignment of property rights to ship ballast waters (see van den Bergh *et al.* 2002). However, the public value of biological resources, such as existence and moral values, cannot be internalized in the market price through the provision and enforcement of property rights, thus hindering the effectiveness of this strategy.

Independent of the policy instrument used, direct government intervention often involves administrative costs, for instance the government may have to establish a monitoring and enforcement agency, thus hampering the effectiveness of this strategy. In addition, the strategy may also be ineffective in the presence of important information asymmetries. This is because information plays a crucial role in the design of an effective Pigouvian tax, particularly when firms have an incentive to conceal true information. In the literature we distinguish two types of informational problems, i.e., hidden information and hidden action problems. Hidden action refers to a post-contractual problem in which one party knows more about his or her type, or effort, after the contract is signed. This is also known as an adverse selection problem. Hidden information refers to a pre-contractual problem, in which one party knows more about his or her true type than the other party before the contract is signed. This is also known as a moral hazard problem – see Akerlof (1970) for a detailed analysis.

Finally, public policies based on direct government intervention may create bureaucratic inefficiency. Bureaucrats may pursue rents and are prone to influences from lobbying activities by market participants (see Milgrom 1988). In fact, in the absence of rent-seeking behavior, such direct government involvement may instead create a disincentive for market participants to innovate or to employ the most efficient method of production.

10.2.2 Information provision

Information provision is an integral part of a public policy directed to the use of market forces without direct government involvement in supply and demand forces. In such a context, policy instruments based on market creation mechanisms have proven to be a valid alternative to direct market intervention policies. The provision of information works within one of two basic conceptual frameworks: certification or ecolabeling. Biodiversity ecolabeling refers to the act of providing information to the consumer that a product, or a product's attributes, possesses specific characteristics regarding the product's origins or ecological, social and economic specifications. Biodiversity certification refers to the act of provision of information with respect to alternative management systems, based on the ability to create a product in an environmentally sound and sustainable manner. Assessing the integrity of a product, or a product's attributes, and the underlying management system involves an evaluation of management practices with respect to defined standards, generally fixed at the management unit level.

In both cases, a credible scheme must evaluate the integrity of the producer's claim and the authenticity of product origin. The assessment of the authenticity of the product's origin involves the identification and monitoring of the supply chain, including raw materials transport and processing, secondary manufacturing and, finally, retail distribution. Therefore, the success of certification and ecolabeling may prove to be difficult to achieve. This strategy often goes hand-in-hand with direct government intervention-oriented public policies, giving rise to 'mixed policies'. The goal of this policy is to circumvent the weaknesses and inefficiencies that may occur when adopting either the command-and-control policy or the market mechanism approach and therefore to achieve higher policy effectiveness. In the next section we explain the use of the certification and ecolabeling policies for conservation and sustainable use of biodiversity and how they can be combined with other instruments to improve their effectiveness.

10.3 Biodiversity certification and ecolabeling as biodiversity policies*

10.3.1 Motivation

Biodiversity certification and ecolabeling refers to an act of provision of information to the consumer about a product, or product characteristics, creating a separate market segment for the product. The participation of consumers in markets for these differentiated products usually permits the market price internalization of some biodiversity benefits. As a matter of fact, consumers are willing to pay a price premium for these benefits. According to the Environmental Protection Agency, several market surveys indicate that a majority of consumers consider themselves to be environmentalists and would prefer to buy products with a reduced environmental impact when the quality and costs are comparable (EPA 1993). The question is then: can consumers, who purchase biodiversity-friendly products, internalize the full range of benefits of biodiversity? As we can see from Figure 10.1, the answer to such questions plays a crucial role in the selection of an effective, broadly accepted biodiversity policy.

If the answer to this question is yes, then biodiversity certification and ecolabeling can effectively create a segment of the market that is a market for biodiversity friendly products

* The authors recognize the contribution of Y.E. Riyanto to this section.

and services. An organic vegetable product, i.e., a vegetable product that is planted without chemical fertilizers, could be an example of such a market segment (Van Ravenswaay 1995; 1996). The underlying steering engine for the creation of such a market relies on the fact that consumers believe that there is a difference in taste between organic and non-organic vegetables. It is often argued that organic vegetables taste better than non-organic ones. In addition, consumers believe that organic products are healthier than non-organic ones and, most of the time, they are able to distinguish between the two products by their appearance. In this setting, the role of certification and ecolabeling is to inform and provide assurance to consumers. Hence, consumers are able to internalize the benefits of consuming the good. Therefore, ecolabeling works as an instrument for resolving the standard hidden information problem.

Most of the time, however, the benefits from biodiversity certification and ecolabeling largely accrue to society at large, and are not explicitly internalized by the individual consumer who purchases the good. In this setting, where benefits from biodiversity certification and ecolabeling are characterized by a public good nature, it is harder to achieve an effective biodiversity certification policy. Figure 10.1 shows that three important factors determine the success of this type of policy. These are consumer awareness, firm incentives to undertake certification and ecolabeling, and the sensitivity of consumer demand to production costs.

10.3.2 Consumer awareness

Consumers' awareness is a necessary condition for the creation of an effective policy certification of public biodiversity benefits. In the extreme scenario of 'no consumer awareness' about environmental and biodiversity protection benefits, which are indirect for the consumption and use of the goods and services, there will be no willingness to pay, or price premium, for such biodiversity benefits (see the study done by Salim *et al.* 1997). Once there is sufficient consumer awareness about the need for a biodiversity-friendly environment, there will be a significant willingness to pay a price premium for ecolabeled or certified products. However, consumer awareness may take many years to develop (see van Ravenswaay and Blend 1997). Hence, policymakers may want to launch extensive information campaigns, targeting the general public, as well as initiate formal education programs about the benefits of having a clean and biodiversity-protected environment (see Figure 10.1).

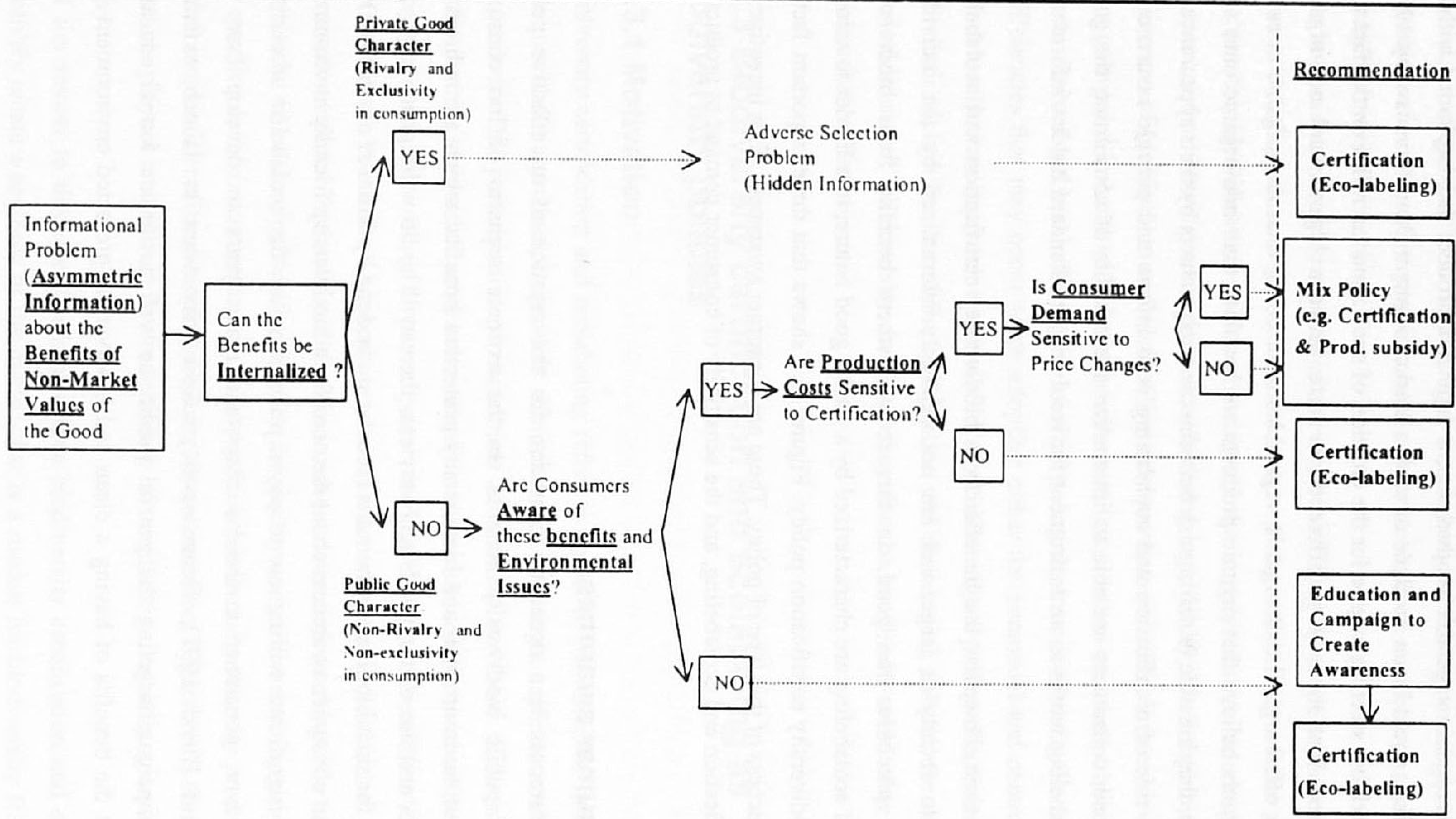

Figure 10.1: A comprehensive view of certification and ecolabeling policies

Source: Nunes and Riyanto (2001).

10.3.3 The incentive of firms to undertake certification and ecolabeling

If firm costs are not sensitive to the costs of undertaking certification and ecolabeling regimes, then producers may have sufficient incentive to incorporate such policies. However, in most cases the adoption of certification and ecolabeling policies would increase firms' production costs because producers may have to install new production technologies or may have to utilize certain inputs in order to satisfy the environmental standards that are stipulated by the product label – see van Ravenswaay and Blend (1997) for more details. Therefore, adopting certification implies incurring higher production costs. These, in turn, damage the firm's market competitiveness, eventually leading to reductions in the firm's profits. Therefore, hardly any producers would like to adhere to certification and ecolabeling regimes. In other words, there are simply not enough market mechanism incentives to make the adoption of certification and ecolabeling policies successful. In such a setting, policymakers may need to complement (or combine) certification policy instruments with other policies aimed at providing enough incentives for producers to adopt certification and ecolabeling. In other words, policymakers may need to launch mixed public policies. For example, it may be effective to combine biodiversity certification and ecolabeling with input subsidies, technical assistance provision, and R&D subsidy regimes (see Figure 10.1).

It is worthy of note that in situations where firms' production costs do not change with the adoption of a certification or ecolabeling regime, it does not mean that a certification or ecolabeling policy is always advisable. In this setting, two markets co-exist, i.e., the market for the conventional product and the market for the certified product. Mattoo and Singh (1994) show that market complementarity between certified and non-certified products can stimulate investment in the technology of the non-certified, or conventional, products – a so-called 'benefit spillover' to the non-certified products. This may lead, in turn, to an increase in the output of the conventional product, in contrast with the original goal of the certification and ecolabeling policy, i.e., gradually increasing the market share of the environmentally friendly product. To avoid such a situation, policymakers are advised to implement certification and ecolabeling schemes together with other public policies, such as the introduction of environmental quality standards and only awarding certificates and ecolabels to those who meet the standards – see Dosi and Moretto (1998) for additional details.

10.3.4 Sensitivity of consumer demand for biodiversity price premiums

If consumers are not willing to pay a premium for certified and ecolabeled products while, at the same time, the introduction of such products boosts firms' production costs, then producers' profits will inevitably decrease. Without any further developments, the certification and ecolabeling regimes would be predestined to fail. In this context, the success of the biodiversity certification and ecolabeling requires that it be combined with other policy strategies. Once again policymakers may want to launch a mixed policy. For instance, policymakers can introduce a certification regime followed by a strong environmental information campaign, stressing the benefits of biodiversity certified products. NGOs also have an important role in building up consumer awareness and disseminating information, for instance through the design and content of ecolabels and certificates. It should be clear to consumers what benefits they can obtain from buying ecolabeled products.

In any situation where consumers are aware of biodiversity benefits, are willing to pay a price premium to consume and use them and the firms' production costs are not too sensitive to the adoption of certification and ecolabeling schemes, policymakers can rely on certification and ecolabeling as effective instruments for protecting biodiversity products, without having the need to combine these instruments with other public policies.

10.3.5 Illustrations of certification, ecolabeling and mixed policy instruments

10.3.5.1 The European organic food market

The European Common Agriculture Policy aims at, inter alia, the reduction of agriculture's environmental impact, and at guiding farmers along the path towards sustainable development practices such as organic farming and integrated crop management. In the last decade all European countries have shown a growth of organic acreage and the number of organic farms. There are, however, considerable variations in such figures between different European countries. In simple terms, four groups of countries can be distinguished: (1) booming countries (Denmark, Finland and Italy); (2) stabilizing countries (Austria, Germany and Sweden); (3) countries with high potential (Greece, Ireland, Norway, Portugal and Spain), and (4) countries lagging behind (Belgium, France, Luxembourg, the Netherlands and the UK). These differences are associated with a wide range of factors such as the diversity of national labor markets, variation in consumer awareness of ecological issues, the distinct direct government interventions in the market's supply and demand forces, and the various labeling and certification strategies for the markets of organic products. The latter implies the use of clear and accurate information on the organic status of the product. For example, when the full standards requirements have been fulfilled, i.e., at least 95 per cent of the ingredients are of certified organic nature, products may be labeled as 'certified organic'. Alternatively, when less than 95 per cent but not less than 70 per cent of the ingredients are of certified organic origin, products can be called 'made with organic ingredients' with a clear statement of the proportion of organic ingredients. Finally, when less than 70 per cent of the ingredients are of certified organic origin, the product may not be called 'organic' (IFOAM 2000). In practice, the creation of an efficient organic food market requires the establishment of a common and reliable chain of products across different European countries. This can be achieved by individual, separate firms for organic products, such as local organic butchers and health food stores, together with a clear, uniform and transparent labeling system under the principles and guidelines of an accredited, independent third-party European authority.

10.3.5.2 The Dutch green-certification system for the energy market

In the Netherlands, a system of green certification was introduced into the energy market starting on 1 January 1998. The aim of the system was to increase Dutch consumption of renewable energy to 10 per cent of total energy consumption by 2020, and to induce producers to increase production of renewable and biodiversity-friendly energy – see Nielsen and Jeppesen (2000) for more details. Because the system was introduced only recently, it is hard to evaluate its performance. Nevertheless, there are two main obstacles to the proper functioning of the green-certification policy, namely the credibility of the certification and the determination of the price ceiling for the green-certified energy. Therefore, government should ensure that if consumers and distributors cannot fulfill the energy quota, then they are required to pay a non-compliance fee or to buy green-certification energy at a higher price. Furthermore, government should ensure that those who break the rule be sufficiently punished.

The price ceiling for the greenlabels, i.e., highest price consumers are willing to pay for them, is the private information of consumers. Therefore, if the government sets a price that is too low, there will not be enough incentive for producers to invest in renewable energy products. On the contrary, if it is set too high, consumers will not individually find it rational to engage in the green-label market, and if they are required to purchase the green labels anyway, will be worse off. In a world of asymmetric information, in which firms have an incentive to conceal true information, it is unavoidable that the government should intensely monitor firms' conduct. Environmental auditing is an example of such monitoring activity.

10.3.5.3 The timber certification market

Timber certification is a process that results in a written statement attesting to the origin of raw wood material, its status and its qualifications. Timber certification typically includes two main components: certification of sustainability of the forest management system, and timber product certification. In simple terms, certification of the forest management system covers forest inventory, management planning, harvesting, road construction and other related activities, as well as the environmental, economic and social aspects of forest activities. Alternatively, timber certification can also be used to validate any type of environmental claim made by the producer, or to provide objectively stated facts about the market system, the timber products and their forest of origin, which normally are not disclosed by the producer or manufacturer (Barron 1994; Bourke 2000a, b; Baharuddin 2001). Two different schemes of timber certification have emerged. On the one hand, there are the principles, guidelines and criteria for accreditation set by the Forest Stewardship Council (FSC) system (e.g., SGS Qualifor, SCS, Rainforest Alliance, Soil Association). On the other hand, there is the International Organization for Standardization (ISO) system and its 1400 series standards relating to environmental management tools and systems to measure a company's practices. There is a consensus that these schemes complement each other. However, FCS is largely supported by NGOs whereas ISO accreditation is perceived to be heavily influenced by the industrialist lobby. The FSC has recently reported that about 17.7 million ha have been certified by FSC-accredited certifiers (FSC 2000). This represents about 0.5 per cent of the world's forest area. Little of this refers to tropical area. As a matter of fact, about 86 per cent of the area certified is in temperate countries, largely Europe and North America. In addition, a new European certification process, the Pan-European Forest Certification Framework, has been launched, with governing bodies established in countries like Austria, Belgium, the Czech Republic, France, Finland, Ireland, Norway, Portugal, Spain, Sweden and Switzerland. In addition Southeast Asian countries have established a national set of criteria for auditing forest management on logging concessions, as well as for the ecolabeling of those concessions' products in light of the International Tropical Timber Organization guidelines – see Nunes and Riyanto (2001) for additional information.

10.3.6 Evaluation of biodiversity certification and ecolabeling policy instruments

It can be concluded that the success of certification and ecolabeling as policy instruments for the creation of markets for biodiversity benefits, which is a crucial tool for protecting biodiversity products and service flows, depends on several crucial factors. These include the ability of the proposed policy instrument to internalize a wide range of the biodiversity benefits, which ultimately depend on the public good nature of the biodiversity benefit under consideration. In addition, three other factors determine the success of biodiversity certification and ecolabeling policy instruments. These are consumer awareness, firm incentives to undertake certification and ecolabeling, and the sensitivity of consumer demand to production costs. From this discussion, it emerges that certification and ecolabeling policy instruments alone are not sufficient to guarantee the successful protection of biodiversity products and services flows. The Dutch energy market, which includes green-energy certification and direct government intervention in the market forces, shows the crucial significance of combining public policies in order to create an effective means of protecting biodiversity products.

Finally, the certification schemes need to be sufficiently flexible to allow for mutual recognition among the agents involved, to meet the demands of weak and sensitive markets, and to avoid encouraging unfair international trade. To achieve this, it is important to explore each country's unique environmental and cultural characteristics. Through mutual understanding and learning from the past, certification and ecolabeling can positively contribute to the creation of markets for biodiversity and thus are expected to assist in the development of effective and broadly accepted sustainable management policies for scarce natural resources.

Chapter 11
CONCLUSIONS

How can we use the ideas presented here to formulate an integrated, effective framework to assess the value of biodiversity? And what can we learn from the large number of available studies? The answers to these questions require, inter alia, that a clear life diversity level be chosen, that a concrete biodiversity change scenario be formulated, that biodiversity changes – notably loss – be within certain boundaries, and that the particular perspective on biodiversity value be made explicit.

So far, most studies lack a uniform and clear perspective on biodiversity as a distinct, univocal concept. In addition, at present we have insufficient knowledge about, for example, how many species there are; for this reason alone, it is very difficult, if not impossible, to assess the total economic value of biodiversity. To completely answer the question, 'What is the value of biodiversity?', we would have to include the value of genetic variation within species across populations, the value of the variety of interrelationships in which species exist in different ecosystems, and the functions among ecosystems. Without any doubt, full monetary assessment is impossible or would be subject to much debate. An additional problem is that, at the global level, biodiversity values can differ significantly, even for similar entities, due to unequal international income distribution. All in all, the available economic valuation estimates should be considered, at best, as a lower bound to an unknown value of biodiversity, and are always contingent upon the available scientific information as well as their global socio-economic context.

As we have seen, biodiversity can be dealt with at different levels: genetic, species, ecosystem, and functional diversity. For the analysis and valuation of biodiversity at the ecosystem and functional levels, which may be regarded as the cornerstone of the analysis and valuation of biodiversity, an active interdisciplinary dialogue is necessary, with emphasis on the complex interface between natural science and social science approaches. A comprehensive assessment of ecosystem biodiversity characteristics, structure, and functioning requires the analyst to take various important steps. First, the socio-economic causes and consequences of biodiversity degradation or loss should be determined. Second, the negative impacts on biodiversity caused by human activities should be assessed. The range and degree of biodiversity functioning should be estimated, especially in terms of ecosystem-functional relationships. Finally, alternative biodiversity management strategies should be ranked and a joint spatial and temporal systems analysis of each policy scenario should be carried out.

The physical assessment of the functions performed by biodiversity is an essential prerequisite of any ecological evaluation. However, simply identifying these functions is insufficient if we want to present resource managers and policymakers with relevant policy response options. It is necessary to develop criteria for the expression of the functions in a form that allows for evaluation. For example, one can identify the range of biodiversity management strategies by exploring the use of methods such as Red Data Species Lists and biological value indexes. Recently, computer models have become available to aid making decisions about species conservation. Models have been applied to calculate minimum dynamic areas that support the minimum viable population of a certain species. In addition, computer models have been used for habitat evaluation, predicting ecological conservation values under different development scenarios. This approach to ecological evaluation allows for a direct comparison of management or conservation strategies.

From an economic perspective, certain aspects of biodiversity are scarce and highly desirable, which is the reason why they have economic value. The concept of economic value is founded in welfare economics, which developed around the theory of consumer behavior. Economic valuation assumes interaction between a subject – a human being – and an object – for example, an element of biodiversity. As a result, economic value is distinct from the notion of intrinsic value, which assumes that an object has or can have value in the absence of any (human) subject. It is important to recognize that economists do not pursue absolute value assessment of environmental systems or all the biodiversity they contain, but always focus attention on valuing environmental system changes. This means that the terms 'economic value' and 'welfare change' are two sides of the same coin. Economics can thus assess the human welfare significance of biodiversity changes, namely through the determination of changes in the provision of biodiversity-related goods and services and their consequent impacts on the well-being of humans who derive – use or nonuse – benefits from their provision. Further note that, although many economic studies employing monetary valuation claim to have assessed biodiversity values, they often confuse biodiversity with biological resources.

Different instruments are available for assessing the monetary value of biodiversity. Stated preference methods have often been used, because the use of revealed preference methods would leave out important biodiversity value types, notably nonuse and quasi-option values. This can lead to a significant value measurement bias, especially since biodiversity conservation is associated with many nonuse and indirect use or primary ecological values. Alternatively, researchers can combine distinct valuation techniques. Special attention, however, should be given to the aggregation of resulting values so as to avoid double counting. From our review of economic valuation studies, it is clear that the assessment of biodiversity values does not lead to an unambiguous monetary value of biodiversity. Instead, available economic valuation estimates should generally be regarded as providing a very incomplete perspective on, and at best lower bounds to, the unknown value of biodiversity changes.

Integrated economic-ecological modeling can contribute to, and may even be essential for, a thorough understanding of the intricate relationship between biodiversity and ecosystem and economic dynamics. Although integrated modeling has somewhat of a tradition, both at the ecosystem level and at the global level, applications to biodiversity-related problems are scarce. Integrated modeling can be linked to biodiversity valuation and evaluation in various ways. Integrated models can generate a set of biological and economic, possibly monetary, indicators that can be further aggregated through multicriteria analysis techniques. In addition, it is possible to provide for a closer, innovative connection between modeling and valuation, among other methods, by: generating conditional values for specific environment-economic scenarios; using scenario-modeling outcomes such as tables and graphs in valuation experiments (e.g., contingent valuation); and using spatial models to aggregate monetary values related to specific areas.

Biodiversity public policy entails the use of direct market intervention, including taxes and command and control instruments, the provision of information, such as certification and ecolabeling, or the combination of both in some sort of policy mix. The success of certification in internalizing biodiversity benefits in the market prices of goods and services, which means that it constitutes an effective instrument in protecting biodiversity, depends on three factors: the public good nature of nonmarket biodiversity benefits; the application of economic valuation methods to assess the monetary magnitude of nonmarket biodiversity benefits; and, supply and demand factors, which include the level of consumer awareness and sensitivity to certain products and the producer's propensity to embrace certification schemes. In some cases certification and ecolabeling policy instruments alone are insufficient to guarantee the successful internalization of biodiversity benefits and thus to contribute to a better allocation of biodiversity. Indeed, mixed policy strategies involving both certification and direct government intervention in supply or demand can sometimes be required.

Finally, one needs to be aware of the limitations of biodiversity valuation and analysis. Biodiversity is a complex and abstract concept. It can be associated with a wide range of benefits to human society, most of them still poorly understood. In general terms, the value of biodiversity can be assessed in terms of its impact on the provision of inputs to production processes, on human welfare, and on the regulation of ecological functions. A complete understanding of these and their integration into multidisciplinary studies provides a great challenge for future research, in which economists, ecologists and others will have to work closely together. Only then can useful policy insights be expected. There is no doubt that the ecological economics of biodiversity will face many research and policy challenges in the years ahead of us.

APPENDIX: RED BOOK CLASSIFICATION OF SPECIES

The following classification was prepared by the IUCN Species Survival Commission and approved during the 40th meeting of the IUCN Council, Gland, Switzerland, 30 November 1994. The list is available at IUCN's Web Page (IUCN 2002).

Critically endangered species

(A) Population reduction in the form of either of the following:

(1) An observed, estimated, inferred or suspected reduction of at least 80 per cent over the last ten years or three generations, whichever is longer, based on (and specifying) any of the following:

(a) direct observation

(b) an index of abundance appropriate for the taxon

(c) a decline in area of occupancy, extent of occurrence and/or quality of habitat

(d) actual or potential levels of exploitation

(e) the effects of introduced taxa, hybridization, pathogens, pollutants, or parasites.

(2) A reduction of at least 80 per cent, projected or suspected to be met within the next ten years or three generations, whichever is the longer, based on (and specifying) any of (b), (c), (d) or (e) above.

(B) Extent of occurrence estimated to be less than 100 km^2 or area of occupancy estimated to be less than 10 km^2, and estimates indicating any two of the following:

(1) Severely fragmented or known to exist at only a single location.

(2) Continuing decline, observed, inferred or projected, in any of the following:

(a) extent of occurrence

(b) area of occupancy

(c) area, extent and/or quality of habitat

(d) number of locations or subpopulations

(e) number of mature individuals.

(3) Extreme fluctuations in any of the following:

(a) extent of occurrence

(b) area of occupancy

(c) number of locations or subpopulations

(d) number of mature individuals.

(C) Population estimated to number less than 250 mature individuals and either:

(1) An estimated continuing decline of at least 25 per cent within three years or one generation, whichever is longer, or

(2) A continuing decline, observed, projected, or inferred, in numbers of mature individuals and population structure in the form of either:

(a) severely fragmented (i.e., no subpopulation estimated to contain more than 50 mature individuals)

(b) all individuals are in a single subpopulation.

(D) Population estimated to number less than 50 mature individuals.

(E) Quantitative analysis showing the probability of extinction in the wild is at least 50 per cent within ten years or three generations, whichever is longer.

Endangered species

(A) Population reduction in the form of either of the following:

(1) An observed, estimated, inferred or suspected reduction of at least 50 per cent over the last ten years or three generations, whichever is longer, based on (and specifying) any of the following:

(a) direct observation

(b) an index of abundance appropriate for the taxon

(c) a decline in area of occupancy, extent of occurrence and/or quality of habitat

(d) actual or potential levels of exploitation

(e) effects of introduced taxa, hybridization, pathogens, pollutants or parasites.

(2) A reduction of at least 50 per cent, projected or suspected to be met within the next ten years or three generations, whichever is longer, based on (and specifying) any of (b), (c), (d), or (e) above.

(B) Extent of occurrence estimated to be less than 5,000 km^2 or area of occupancy estimated to be less than 500 km^2, and estimates indicating any two of the following:

(1) Severely fragmented or known to exist at no more than five locations.

(2) Continuing decline, inferred, observed or projected, in any of the following:

(a) extent of occurrence

(b) area of occupancy

(c) area, extent and/or quality of habitat

(d) number of locations or subpopulations

(e) number of mature individuals.

(3) Extreme fluctuations in any of the following:

(a) extent of occurrence

(b) area of occupancy

(c) number of locations or subpopulations

(d) number of mature individuals.

(C) Population estimated to number less than 2,500 mature individuals and either:

(1) An estimated continuing decline of at least 20 per cent within five years or two generations, whichever is longer, or

(2) A continuing decline, observed, projected, or inferred, in numbers of mature individuals and population structure in the form of either:

(a) severely fragmented (i.e., no subpopulation estimated to contain more than 250 mature individuals)

(b) all individuals are in a single subpopulation.

(D) Population estimated to number less than 250 mature individuals.

(E) Quantitative analysis showing the probability of extinction in the wild is at least 20 per cent within 20 years or five generations, whichever is longer.

Vulnerable species

(A) Population reduction in the form of either of the following:

(1) An observed, estimated, inferred or suspected reduction of at least 20 per cent over the last ten years or three generations, whichever is longer, based on (and specifying) any of the following:

(a) direct observation

(b) an index of abundance appropriate for the taxon

(c) a decline in area of occupancy, extent of occurrence and/or quality of habitat

(d) actual or potential levels of exploitation

(e) the effects of introduced taxa, hybridization, pathogens, pollutants, or parasites

(2) A reduction of at least 20 per cent, projected or suspected to be met within the next ten years or three generations, whichever is longer, based on (and specifying) any of (b), (c), (d) or (e) above.

(B) Extent of occurrence estimated to be less than 20,000 km^2 or area of occupancy estimated to be less than 2,000 km^2, and estimates indicating any two of the following:

(1) Severely fragmented or known to exist at no more than ten locations.

(2) Continuing decline, inferred, observed or projected, in any of the following:

(a) extent of occurrence

(b) area of occupancy

(c) area, extent and/or quality of habitat

(d) number of locations or subpopulations

(e) number of mature individuals.

(3) Extreme fluctuations in any of the following:

(a) extent of occurrence

(b) area of occupancy

(c) number of locations or subpopulations

(d) number of mature individuals.

(C) Population estimated to number less than 10,000 mature individuals and either:

(1) An estimated continuing decline of at least 10 per cent within ten years or three generations, whichever is longer, or

(2) A continuing decline, observed, projected, or inferred, in numbers of mature individuals and population structure in the form of either:

(a) severely fragmented (i.e., no subpopulation estimated to contain more than 1,000 mature individuals)

(b) all individuals are in a single subpopulation.

(D) Population very small or restricted in the form of either of the following:

(1) Population estimated to number less than 1,000 mature individuals.

(2) Population is characterized by an acute restriction in its area of occupancy, typically less than 100 km^2, or in the number of locations, typically less than five. Such a taxon would thus be prone to the effects of human activities, or stochastic events whose impact is increased by human activities, within a very short period of time in an unforeseeable future, and is thus capable of becoming Critically Endangered or even Extinct in a very short period.

(E) Quantitative analysis showing the probability of extinction in the wild is at least 10 per cent within 100 years.

(b) area of occupancy

(c) number of locations or subpopulations

(d) number of mature individuals.

(C) Population estimated to number less than 10,000 mature individuals and either:

(1) An estimated continuing decline of at least 10 per cent within ten years or three generations, whichever is longer, or

(2) A continuing decline, observed, projected, or inferred, in numbers of mature individuals and population structure in the form of either:

(a) severely fragmented (i.e., no subpopulation estimated to contain more than 1,000 mature individuals)

(b) all individuals are in a single subpopulation.

(D) Population very small or restricted in the form of either of the following:

(1) Population estimated to number less than 1,000 mature individuals.

(2) Population is characterized by an acute restriction in its area of occupancy (typically less than 100 km²) or in the number of locations (typically less than five). Such a taxon would thus be prone to the effects of human activities, or stochastic events whose impact is increased by its vulnerability within a very short period of time in an unforeseeable future, and is thus capable of becoming Critically Endangered or even Extinct in a very short period.

(E) Quantitative analysis showing the probability of extinction in the wild is at least 10 per cent within 100 years.

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